

# EXHIBIT 7

## **Declaration of Graham Gagnon, Ph.D., P.Eng.**

Armview Engineering Ltd.

*In Re* Flint Water Cases, U.S. District Court for the Eastern District of Michigan, Civil  
Action No. 5:16-cv-10444-JEL-MKM (coordinated docket)

*Carthan, et al. v. Snyder, et al.*, Civil Action No. 5:17-cv-10164-JEL-MKM

January 6, 2021

## 1 Scope of Engagement

I was asked by Campbell, Conroy, & O'Neil, P.C. on behalf of Veolia North America ("VNA") to provide my opinions about water chemistry and treatment issues relevant to the City of Flint, Michigan's use of the Flint River as a public drinking water source in 2014 and 2015. Specifically, I was asked to evaluate sources of lead in drinking water, NOM/TOC-related water quality and treatment issues, and corrosion control for treated Flint River water delivered to consumer taps after April 2014.

My opinions expressed in this report are offered within a reasonable degree of scientific certainty, and are based on my training, experience, and knowledge in the broad field of environmental engineering with an emphasis on water treatment. I understand that discovery is continuing, and additional evidence may become available. I reserve the right to amend my opinions and report if additional information is received.

For this engagement, I have been compensated at a rate of \$280 per hour for preparation and \$420 per hour for testimony. In the course of forming my opinions and writing this report, I have reviewed and relied upon materials from the litigation, including deposition transcripts and exhibits, as well as non-litigation materials, including those cited in this report and listed below as references.

## 2 Expert Qualifications

I am a Full Professor in the Department of Civil & Resource Engineering at Dalhousie University. I am the NSERC/Halifax Water Industrial Research Chair in Water Quality & Treatment and the Director for the Centre for Water Resources Studies at Dalhousie. I am a licensed Professional Engineer in the province of Nova Scotia and the President of Armview Engineering Ltd.

I hold a PhD in Civil Engineering from the University of Waterloo and a bachelor's degree in Environmental Engineering from the University of Guelph. For over 25 years, I have conducted research in drinking water quality and treatment, with an emphasis in distribution system water quality. I have published over 175 peer-reviewed journal articles, over 300 conference presentations and contributed to more than 20 refereed books or research reports. My research contributions have addressed lead, copper and iron corrosion in drinking water systems; evaluation of corrosion control agents in drinking water; optimization and assessment water treatment processes; biofilm disinfection in water systems; sampling and measurement of drinking water parameters; and many other water treatment related topics. I have conducted research in controlled laboratory environments, in the field using pilot-based equipment (such as pipe loops, annular reactors, pilot-water treatment plants) and sampling from full-scale water treatment plants and distribution systems. I have worked on drinking water research projects with utilities across Canada and the United States.

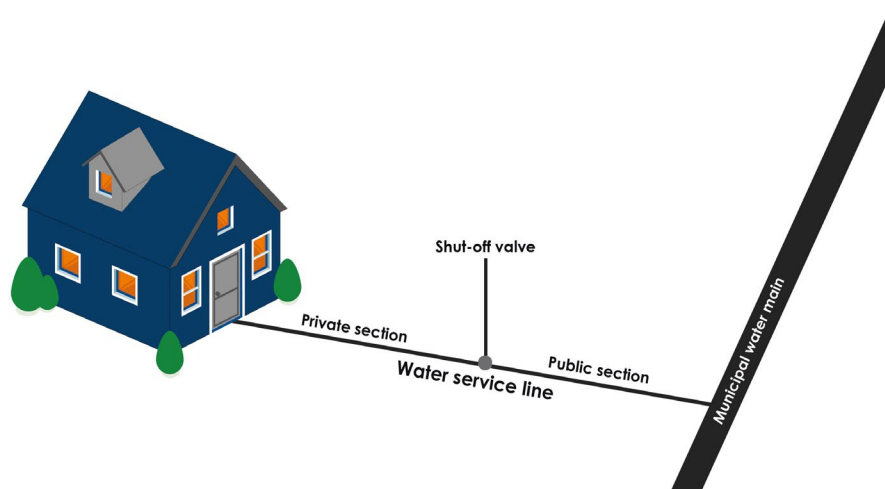
### 3 Opinions

#### 3.1 Sources of Lead in Flint Drinking Water

Testing at Flint's water treatment plant in 2014-15 consistently showed no detectable lead in water drawn from the sample tap in the plant, per the utility's reporting to Michigan's Department of Environmental Quality (MDEQ). That is, lead was not present in treated water before it entered Flint's drinking water distribution system. The potential sources of lead contamination in Flint were thus the distribution system and its associated components (e.g., water mains, service lines, etc.) and/or premise plumbing in individual households.

A distribution system is the portion of a water system that transports water from the treatment plant to the customer. Drinking water distribution systems include pipes called water mains that carry water from a centralized treatment plant throughout the distribution network. Water mains are commonly made of iron, as they were in Flint, or of steel. They do not contain lead.

Individual houses are connected to mains by smaller diameter pipes called service lines. Service lines vary in composition: lead, copper, galvanized iron, or PVC are typical. Typically, homeowners are responsible for the portion of the service line on their property (private section), while municipalities or water utilities are responsible for the portion up to the property line (public section) (Figure 1).



**Figure 1.** *The water service line (Montreal West, 2019).*

The service line connects to premise plumbing, which is pipe material inside the home or dwelling. Premise plumbing contains various types of valves and shutoffs, hose bibs, faucets, and appliances such as hot water heaters, washing machines, and dishwashers. Lead may be present at very low levels within premise plumbing, as either fittings, solder, or fixtures, which contain relatively small amounts of lead.

Historically in the United States, lead was commonly used as a service line material. It is estimated that more than six million homes have some form of a lead service line (Cornwell et al., 2016). However, as discussed below, only a small fraction of homes in Flint had lead service lines between 2014 and 2016. It is the broad collective opinion of many agencies, including the American Water Works Association Research Foundation (Sandvig et al., 2008) and the American Water Works Association (AWWA, 2020), that lead service lines represent a significant source of lead in drinking water.

Given the public health concerns surrounding elevated lead in drinking water, the removal and replacement of lead service lines has become an important strategy for reducing lead exposure from drinking water (Trueman et al., 2016). However, when a lead service line is connected to a pipe made of a different metal, a partial lead service line will exist. Partial lead service lines have been shown to be very problematic in the short term (months after replacement), and in the long-term (legacy) lead release from partial replacements has been shown to be comparable to no replacement (full lead service line) (Trueman et al., 2016).

### **3.1.1 Understanding Lead Service Lines through the FAST Start Program**

The presence of lead service lines in Flint is now well documented (e.g, Roy and Edwards, 2020). To understand the incidence and ownership of lead service lines in Flint, Armview Engineering reviewed data generated by the Flint Fast Action and Sustainability (FAST) Start program in connection with a project intended to replace the main source of lead in Flint's water distribution system: lead service lines.

As described by Abernathy et al. (2018): "the Michigan state legislature eventually appropriated \$27M towards the expensive process of replacing these lines at large scale; later the U.S. Congress allocated another nearly \$100M towards the recovery effort. The group directed to execute the replacement program was called Flint Fast Action and Sustainability program (FAST Start), and their task was to remove as many hazardous service lines as possible up to funding levels."

The FAST Start program was designed as a targeted program to remove known or suspected lead service lines from properties in Flint. A survey of service lines identified that the City of Flint has over 56,000 service lines with the majority being copper and smaller fraction being lead (Rowe 2016). Service lines have been extracted from Flint through the FAST Start program from 2016 to 2020. The FAST Start program was contracted to AECOM to execute. Based on data from the FAST Start program (through October 14, 2020) the total number of addresses in the FAST Start program was 25,948 (Rosen, 2020). Based on analysis from Rosen (2020), the following information concerning service line materials were obtained:

- There were 9,571 addresses in the FAST Start program that had lead pipe from their property (public or private), which represents 34% of the total addresses.
- There were 19,754 addresses in the FAST Start program that had copper pipe from their property (public or private), which represents 76% of the total addresses.

- There were 4,878 addresses in the FAST Start program that had galvanized pipe from their property (public or private), which represents 19% of the total addresses.

As noted by the City of Flint (2016), “Under FAST Start, crews continue to replace service lines in neighborhoods most likely to have lead service lines, and where a significant number of young children or seniors live.” Therefore based on current data from FAST Start, as few as 17% of service lines in Flint are lead, in whole or in part.

To date, the number of lead service lines extracted from the FAST Start are substantially lower than previous estimates. Abernathy et al. (2016) utilized a machine learning algorithm to estimate that “approximately 29,100 parcels (about 52% Flint parcels) would require some replacement. Among those, about 17,500 would be full replacements and 11,600 would be partial replacements.” Based on subsequently available data, that appears to have been a significant overestimate.

Data in the FAST Start program were organized in terms of pipe material, ownership, and replacement type: full, public partial, or private partial. Public partial refers to the portion of the lead service line that is the responsibility of the City of Flint and Private portion refers to the portion of lead service that is the responsibility of the homeowner (i.e., the owner of the property address).

Based on analysis from Rosen (2020) the majority (8,806 out of 9,571) of estimated lead service lines in the system were publicly owned. In other words, of the lead service lines recovered through the FAST program 92% were owned by the City of Flint. By comparison, 8% (or 412 out of 9,571) were privately owned by the property owners. It is reasonable to expect that the number and ownership of lead service lines in Flint was roughly constant between April 2014 and 2016. There are now even fewer lead service lines remaining in Flint and those that remain are in the process of being replaced. Presently, the public and private lead service lines that remain represent a tiny fraction of the service lines in Flint. The removal of most lead service lines in Flint has resulted in a significant reduction in citywide lead exposure, as described by Roy and Edwards (2020).

### **3.1.2 Lead Service Lines were the Predominate Source of Lead in Flint**

Several studies have concluded that lead service lines were the major source of lead in Flint’s drinking water, including:

- Mantha et al. (2020), who investigated lead concentrations in two sentinel homes in Flint. As reported by the authors “the cumulative mass of Pb released dropped below 1 mg per sampling round after lead service line (“LSL”) replacement, as the replacement removed the massive reservoir of Pb.” Indeed, the authors concluded that “LSL was the major source for Pb, and LSL replacement was the most effective intervention for reducing Pb levels in the long term.”

- Pieper et al. (2018), who found that lead concentrations at the tap in Flint were higher when the data analyzed considered homes with lead service lines. In the context of lead concentrations in Flint, the authors reported that “the average 90th percentile [for Pb] was 18.2 µg/L (±5.2 µg/L) when 0% of homes were LSLs (n = 0 of 30), compared to 31.2 µg/L (±8.1 µg/L) when 50% of homes were LSLs, and 68.4 µg/L (±10.4 µg/L) when 100% of homes were lead pipe.”
- Lytle et al. (2019), who conducted sampling (208 sets of samples from 107 different locations) to identify sources of lead in drinking water and who noted that when present, lead service lines were the greatest source of lead and following their removal, the total mass of lead contributed to the drinking water was reduced by 86% on average.
- Abokifa et al. (2020), who reviewed over 35,000 samples from the Flint system (over 85% were collected through crowdsourced sampling) and analyzed those samples for lead. Results indicated that regions with high water lead levels were consistent with the areas where lead service line density was the highest.
- Finley (2021) evaluated water lead levels from samples collected by Dr. Marc Edwards and co-workers in Flint during 2015 and cross referenced those data with FAST Start service line and replacement data. Finley (2021) was able to statistically determine that “the first draw, one minute flush, and three minute flush samples, the WLLs [water lead levels] for the addresses with either lead public or private service lines were significantly different than the WLLs for the addresses without lead public and private service lines (p<0.00001 for all tests) for the August/September 2015 sampling event.” This analysis demonstrated that addresses with lead service lines had lead concentrations that were significantly greater than those addresses that did not have lead service lines.
- Roy et al. (2019) and Roy and Edwards (2020), who analyzed lead contained within biosolids. Dr. Edwards’ team analyzed metals data that was collected in biosolids (i.e., the solid waste generated during wastewater treatment) from the Flint Water Pollution Control Plant. Dr. Edwards’ team was the first group to apply to water treatment issues an accepted concept of wastewater surveillance or wastewater epidemiology that has been used to understand illicit drug use (e.g., Been et al., 2016) and microbial pathogens such as SARS CoV-2 (e.g., Sherchan et al., 2020). Through Dr. Edwards’ analysis, it was demonstrated that there were peaks in biosolid lead concentrations in 2011 and 2014. Further, there was a decline in biosolids lead concentration in 2019 which was attributed to corrosion control and lead service line replacements in the City of Flint. In particular, Roy and Edwards (2020) stated: “The efficacy of enhanced corrosion control and replacement of service lines that contain lead is, therefore, on the order of 72–84% effective at reducing citywide lead exposure.” At his deposition, Dr. Edwards explained the biosolids analysis by stating that “[i]f you look at how much lead is in the sewage, you can get an idea of how much lead was released to the drinking water every month going back for years,” and that “in this article we proved that concept seemed to apply showing that the lead in the biosolids agreed with random citywide sampling events.” (Edwards Dep., p. 426).

The biosolids work conducted by Dr. Edwards' team revealed that lead concentrations were comparable in 2011 and 2014. This finding speaks to the legacy challenge that lead service lines posed in Flint. Flint's drinking water in 2011 was provided by Detroit Water Sewerage District, sourced from Lake Huron, and had very different chemistry than water treated from the Flint River. Dr. Edwards' biosolids work delineates some factors that may have caused this increase in 2011. However, it is possible that chronic challenges such as water age in Flint's distribution system, iron corrosion from cast iron water mains, and low chlorine residuals, may have contributed.

A recent presentation by David Cornwell through the Water Research Foundation (Evaluation of Flushing to Reduce Lead Levels WRF #4584, August 9, 2018) showed, in general, that the highest lead concentrations were in lead service lines. This particular study draws attention to the fact that lead concentrations within a specific home will be highly variable and dependent on a variety of home-specific conditions. By extension, these home-specific conditions would result in different lead concentration profiles in the Flint properties containing with lead service lines, versus those without lead service lines.

Further, a USEPA presentation on sequential sampling (Third Flint Data Summit – Region 5, January 10, 2017) noted that “[m]ost of the lead [in Flint] is coming from the service lines” and that “particulate lead yields the highest concentration of lead; particulate lead release is random/sporadic.” In the same presentation, the total lead mass reduction in homes post-replacement of lead service lines was generally between 70 and 90% (with exception of two sites with 22 and 50% reduction). When sequential samples were collected, the highest lead concentrations were observed in samples from the lead service line. Consistent with other studies, there is very little evidence that would suggest that premise plumbing was a significant contributor to lead concentrations in Flint. There is very little evidence in the literature that would demonstrate the benefit of replacing premise plumbing to lower lead concentrations at the community level. This presentation also conformed with data from lead monitoring results in Flint, where official Lead and Copper Rule (LCR) sampling (90th percentile first draw lead) revealed a steady decrease in lead concentrations as a result of removal of lead service lines as part of the FAST program. In 2016, lead concentrations in some samples were upwards of 20 ug/L, while as of June 2019 lead concentrations were between 2 and 6 ug/L (State of Michigan, 2020). In addition to implementing improved treatment for NOM removal coupled with corrosion control, the removal of lead service lines resulted in reductions in water lead levels to below the compliance value of 15 ug/L.

The EPA published the Lead and Copper Rule Revisions on December 2020. For systems with lead service lines, the EPA now requires a LSL Replacement (LSLR) plan: “All water systems with LSLs or lead status unknown service lines in their initial inventory must create and submit an LSLR plan to their state by the rule's compliance date. The LSLR plan must include a description of: (1) a strategy for determining the composition of lead status unknown service lines in its inventory, (2) procedures to conduct full LSLR, (3) a strategy for informing customers before a full or partial LSLR, (4) for systems that serve more than 10,000 persons, a recommended LSLR goal rate in the event of a lead trigger level exceedance, (5) a procedure for customers to flush service lines and premise plumbing of particulate lead, (6) a LSLR prioritization strategy based on factors including but not limited to the targeting of known LSLs, LSLR for disadvantaged consumers and populations most sensitive to the effects of lead, and (7) a funding strategy for conducting LSLRs which considers ways to accommodate customers that are unable to pay to



replace the portion they own. Completing a LSLR plan will prepare water systems to take the steps necessary to remove a source of drinking water lead exposure when required.” As noted in item 5 (above), the LSLR plan recommended by EPA requires a strategy flushing of premise plumbing but does not require replacement of premise plumbing following LSL replacement.

### **3.1.3 Release of Lead from Pipe Scale in Flint**

Soluble lead can in theory be released from lead service lines by corrosion processes. Corrosion is an electrochemical reaction that requires the presence of an anode, a cathode, an internal circuit, and an electrolyte. The corrosion process results in the destruction of a metal through redox (reduction-oxidation) reactions (Snoeyink and Wagner, 1996). The type of lead service line replacement can govern the type of corrosion that occurs, as well as the rate of lead release. In a corrosion reaction, oxidation of a metal takes place at the anode, releasing an electron and a positively charged ion which move toward the cathode. Negative ions generated at the cathode tend to migrate towards the anode to maintain neutrality of the solution (Snoeyink and Wagner, 1996).

Uniform and galvanic corrosion are most commonly encountered in water distribution systems. Uniform corrosion occurs on a single metal pipe when areas on the pipe surface transition between anodic and cathodic behavior. In uniform corrosion, the anodes and cathodes constantly shift along the surface scale, resulting in uniform losses along the pipe surface. Uniform corrosion controls lead dissolution from lead service lines. In comparison, galvanic corrosion is a chemical reaction between dissimilar metals. It can cause lead release to drinking water when lead comes into contact with another metal, most often copper (Gregory, 1985; Dudi, 2004; Edwards and Triantafyllidou, 2007). In galvanic corrosion, copper behaves as the cathode (Dudi, 2004), resulting in protection (reduction) of the copper surface, while lead behaves as the anode and is oxidized, resulting in the acceleration of lead release.

However, lead release in Flint was predominantly driven not by uniform or galvanic corrosion, but by lead detaching from scale in the form of tiny particles, and/or dissolving from pipe scale that contained lead and other minerals on the inner surfaces of lead service lines. Williams et al. (2018) conducted an analysis of lead service lines extracted from Flint and found that the lead pipe consisted of three layers of scale above the lead pipe surface. In examining the scales, Williams et al. (2018) described the scale in the top layer (i.e., the layer in contact with water) as being amorphous. The top layer of the scale contained elements such as lead, iron, manganese, and trace amounts of carbon. Table 1 shows the percent by weight of some of the elements contained within that top layer of scale (Layer 1), which would have been in contact with Flint water. Because the top layer was characterized as amorphous, it is accepted that available chemical and thermodynamic equations would not be able to predict lead release from the extracted lead pipe. Given that lead service lines in Flint were over 70 years old (Rowe Engineering, 2013), the pipe scale on lead service lines would have been ubiquitous and thus the amount of “bare” or exposed pipe in Flint would have been negligible.

**Table 1.** Average percent (%) by weight of elements in the uppermost layer (Layer 1) of lead pipe scale from Flint pipe rig study (Williams et al., 2018).

Element	Proportion in Layer 1 Scale (% weight)
Pb	18.03-21.25
Mn	6.39-7.12
Fe	6.13-6.16
C	2.34-2.42

In drinking water, metals are commonly classified and measured as “total,” “dissolved,” or “soluble,” where soluble and dissolved are used interchangeably. In this classification, water samples containing metals and other constituents like NOM are passed through a filter paper with a 0.45 µm pore size. “Dissolved” or “soluble” referred to as the elements that have passed through the 0.45 µm filter paper, while particulate matter is retained on the filter. This analytical procedure is described in EPA Method 200.8, Revision 5.4: Determination of Metals and Trace Elements in Water and Wastes by Inductively Coupled Plasma-Atomic Emission Spectrometry Section 8.2 (USEPA, 2015). Namely, “for the determination of dissolved elements, the sample must be filtered through a 0.45 µm pore diameter membrane filter.” Here, total metal concentrations represent a digestion of the entire water sample that has not been filtered, which would include the particulate metal and the dissolved portion.

In the Williams et al. (2018) paper, a variance from the USEPA Method 200.8, Revision 5.4 (USEPA, 2015) for quantifying dissolved metals was used. As described by Williams et al. (2018):

Two 30 mm syringe filters were used to remove lead particles, depending on availability. Specifically, either 0.1 µm pore-sized nylon filters, or 0.2 µm polypropylene syringe filters were used. Both types of filter units had been tested in prior studies to minimize or limit sorption losses of lead to the filter unit. . . . During the period of June 2016 to March 2017, filtered samples were taken at as many sampling events as possible. The first two sampling events used 30 mm 0.2 µm pore polypropylene syringe filter disks, and subsequent sampling events used 30 mm 0.1 µm pore-sized nylon syringe filter disks.

Thus, the majority of the soluble lead samples in the work by Williams et al. (2018) were collected through a 0.1 µm filter, and therefore the soluble data presented in Williams et al. (2018) is less than 0.1 µm *in size*. The data from the study noted that the particulate lead (particles greater than 0.1 µm) consisted of between 77 and 100% of the total lead present in their study. In other words, nearly all of the lead was present in particulate form.

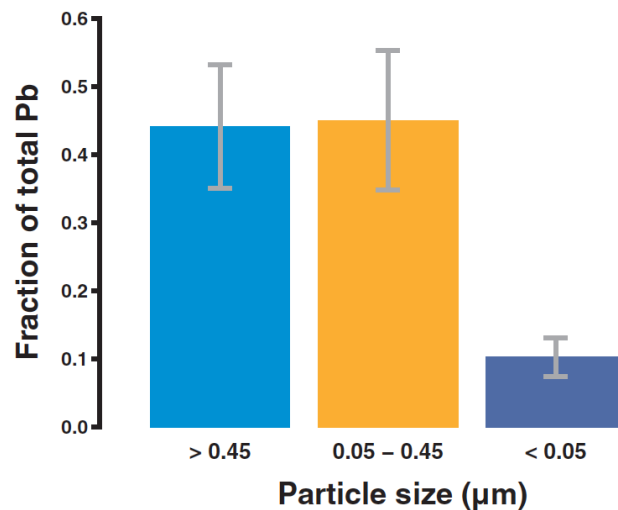
The approach used by Williams et al. (2018) depicts more information on smaller particle size of 0.1 µm lead particles, compared to the USEPA Method 200.8 convention of filtering through a 0.45 µm filter. Specifically, Williams et al. (2018) described “particulate lead (defined as larger than 0.1 µm or 0.2 µm) as the dominant lead release mechanism during the test period, generally accounting for 75% to nearly 100% of the lead release.... [furthermore], lead release is

dominantly and consistently in the form of very small particulates.” While the authors did not provide further quantitative analysis of these “very small particulates,” we know that some of these very small lead particles were as small as 0.1  $\mu\text{m}$  (or 100-nm) and they would have originated from amorphous pipe scale that was enriched with iron, manganese and carbon. For example, Williams et al. (2018) commented that the “porous and poorly adherent properties suggest that the scale is potentially susceptible to lead particulate release during normal use or if disturbed hydraulically or physically.”

In his deposition, Dr. Edwards also confirmed the presence of particulate lead in the Flint system, where the origin was from pipe scale. He noted that “[t]his is ... a case where virtually all of the lead was particulate” and that “So this is not a situation where if I flushed the water I’m down to zero. It’s like I am sloughing off solids from the pipe, that’s the whole mechanism...It’s not a soluble release mechanism where we’d have a bigger argument, okay, about how to weight it. If it was all soluble after the first six or seven liters, you would have no lead in your water.” (Edwards Dep., p. 468-469). As discussed previously, Roy and Edwards (2020) examined the mass of lead in Flint biosolids and found a decline in total lead release due to the removal of service lines as well as the implementation of corrosion control treatment (e.g. addition of orthophosphate). In his deposition, Edwards noted that “[t]here is so much particulate matter in sewage and lead likes to stick to the particles, generally speaking 85 percent or more of the lead, whether it is soluble or particulate, is captured in biosolids, and the lead in Flint, the vast, vast majority of it started as a particle. Virtually all of the lead from [a Flint residence studied by Dr. Edwards], virtually all of the lead we measured in Flint was -- was particulate.” (Edwards Dep., p. 733). This confirms that the lead present in the Flint system was particulate and originated from sloughing off the lead pipe.

To further understand these particles, a methodology was developed by Trueman and Gagnon (2016) in which water that was filtered through a 0.45  $\mu\text{m}$  paper was then passed through a size exclusion chromatography column prior to analysis on ICP/MS. The size exclusion chromatography column separates “very small particles” to understand the characteristics of particles in the size range between 0.05 and 0.45  $\mu\text{m}$ .

Using this analytical procedure, Trueman and Gagnon (2016) found that in drinking water exposed to lead pipe, “a considerable fraction (0.45, std. dev. 0.10) of total lead in samples occurred in the 0.05–0.45  $\mu\text{m}$  size range,” in other words between 35 and 55% of the total lead was between 0.05–0.45  $\mu\text{m}$  in size (Figure 2). For comparison, the Williams et al. (2018) study found 70 to 100% of samples were in the size range greater than 0.1 to 0.2  $\mu\text{m}$ . Both studies point to the fact that the truly dissolved portion of lead is very small, and in fact, lead existed in a particulate form that contains “very tiny particle” which are in the size range 0.1 to 0.45  $\mu\text{m}$  (e.g., 100-450 nm). Specifically, Trueman and Gagnon (2016) showed that approximately 10% or less of the total lead was smaller than 0.05  $\mu\text{m}$  compared to Williams et al. (2018) who found an average of approximately 12% of total lead smaller than 0.1  $\mu\text{m}$  in size. This illustrates how most of the lead were very small particles, as opposed to being soluble or truly dissolved.



**Figure 2.** Mean fraction of total lead in each size range (from Trueman and Gagnon, 2016).

Further, in Trueman and Gagnon (2016), samples were also filtered through a 0.1 µm filter paper and their size distributions were characterized via size exclusion chromatography. The relative metal and NOM (UV254) concentrations were measured as a function of size. This analysis showed that the lead and iron particles in the 0.1 µm size range had a UV254 signal. This means that the lead and iron in the samples had adhered to organic matter. In other words, the particles present consisted of lead and iron as well as NOM. Trueman and Gagnon (2016) explained that “lead occurs in distributed drinking water adsorbed to particles rich in iron and natural organic matter.”

### 3.1.4 Lead Release in Flint was not Predictable through Corrosion Indices

Release of lead solids of the kind described above is not predictable by water quality indices applicable to galvanic corrosion or calcium compound precipitation. Such indices have been developed based on experimental or field scale work to determine whether the water will be corrosive or whether scale will form. The most commonly used indices include the chloride to sulfate mass ratio (CSMR), the Langelier Saturation Index (LSI) and the Larson-Skold Index. However, it is important to note that several agencies as well as the corrosion literature recommend against the use of these indices. Health Canada (2009) noted that “[c]orrosion indices should not be used to assess the effectiveness of corrosion control programs, as they provide only an indication of the tendency of calcium carbonate to dissolve or precipitate.” Furthermore, the American Water Works Association Research Foundation recommended that the use of corrosion indices such as the LSI for corrosion control practices be abandoned (Benjamin et al., 1996). Singly (1981) noted that “no general corrosion index is available” in drinking water distribution systems and that there is little likelihood that a single index can apply to all metal.

The CSMR is the ratio of chloride to sulfate in treated drinking water. The CSMR ratio attempts to assess the corrosivity of a drinking water system when lead, or lead containing materials (primarily solder and leaded brass fixtures), is in direct contact with copper. CSMR has

been used based on observational data to indicate lead release from galvanic connections such as lead solder on copper pipes or partial lead service line replacements (Nguyen et al., 2011).

There is some limited field experience and some laboratory support for a CSMR greater than 0.58 leading to corrosion rates of lead and leaded alloys resulting in lead levels at the tap that exceed the allowable limit for lead in drinking water (Edwards, et. al. 1999). The 0.58 value is empirically determined and has no fundamental basis in corrosion science. As noted by Masten (2016), CSMR is an empirical parameter that has “no sound theoretical underpinning.”

Further, a recent study by Arnold et al. (2020) surveyed water utilities and their corrosion control practices. As noted by Arnold et al. (2020), “[a]pproximately 75% of the systems in the survey had CSMR values greater than 0.5. No significant correlations between reported CSMR values and 90th percentile lead values were observed when evaluating the systems in this study.”

The LSI estimates the theoretical tendency to form calcium carbonate scale, which inhibits corrosion and is based on known chemical equilibria for calcium carbonate (Crittenden et al., 2012). When the LSI is greater than 0, the water will tend to precipitate calcium carbonate scales, whereas when the LSI is less than 0, the water will tend to dissolve calcium carbonate scales.

Although the LSI tends to estimate whether calcium carbonate will precipitate or dissolve, it does not predict how much will precipitate or whether the structure of the precipitated solid will provide resistance to corrosion (Larson, 1975). The LSI is not specific to any metal surface and is based on known chemical equilibria for calcium carbonate (Crittenden et al., 2012). Because of contradictory evidence between corrosion and the use of indices, the American Water Works Association Research Foundation recommended that the use of corrosion indices such as the LSI for corrosion control practices be abandoned (Benjamin et al., 1996).

In 2009, Health Canada promulgated the *Guidance on Controlling Corrosion in Drinking Water Distribution Systems* with the definition of “corrosion control” of controlling the leaching of metals, specifically lead, that results from the corrosion of materials in drinking water distribution systems. Within the Health Canada (2009) guideline, the following statement concerning corrosion indices is presented:

Corrosion indices should not be used to assess the effectiveness of corrosion control programs, as they provide only an indication of the tendency of calcium carbonate to dissolve or precipitate. They were traditionally used to assess whether the distributed water was aggressive towards metals and to control for corrosion. These corrosion indices were based on the premise that a thin layer of calcium carbonate on the surface of a metallic pipe controlled corrosion. Accordingly, a number of semi-empirical and empirical relationships, such as the Langelier Index, the Ryzner Index, the Aggressiveness Index, the Momentary Excess and the Calcium Carbonate Precipitation Potential, were developed to assess the calcium carbonate-bicarbonate equilibrium. However, a deposit of calcium carbonate does not form an adherent protective film on the metal surface. The work of Edwards et al. (1996) has even shown that under certain conditions, the use of corrosion indices results in actions that may increase the release of corrosion by-products. In light of significant empirical evidence contradicting the presumed connection between corrosion and the most common

of the corrosion indices, the Langelier Index, the American Water Works Association Research Foundation recommended that the use of corrosion indices for corrosion control practices be abandoned (Benjamin et al., 1996).

Many water treatment and engineering textbooks that are widely used by industry professionals also question the applicability of corrosion indices. For example, in the MWH textbook “Water Treatment: Principles and Design” by Crittenden et al. (2012), it was noted that “the results summarized above were developed by Stumm (1960) in a study examining the corrosion of cast-iron pipe during exposure to several different water qualities. Based on these results, no correlation was found between the Langelier index ( $\text{CaCO}_3$  saturation) and corrosion even though  $\text{CaCO}_3$  deposition did increase substantially as the Langelier index increased.” (p. 1775). In the textbook *Theory and Practice of Water and Wastewater Treatment* by Droste (1997), there is a section entitled “The Langelier Index Misconception” that “The LI [Langelier Index] is only a qualitative indication of the amount of potential  $\text{CaCO}_3$  deposition...these indices should not be used as indicators of corrosion potential.”

Furthermore, Singley (1981) noted that “no general corrosion index is available” in drinking water distribution systems and that there is little likelihood that a single index can apply to all metal. This perspective on indices was also reinforced in a presentation by Triantafyllidou and Schock (2019), where it was noted that most corrosion indices were not developed or named for corrosion (e.g. saturation, stability, etc.). Specifically, Triantafyllidou and Schock (2019) noted that “the LSI and subsequent variations predict calcium carbonate scaling, not corrosion” and that “only in rare cases is calcium carbonate an active inhibition mechanism” (slide 22). In addition, it was determined that when used outside their limitations, these indices become unreliable, there are different objectives for different materials, and that there is no universal index for multiple materials. Finally, Triantafyllidou and Schock (2019) summarized their presentation by noting that calculating “corrosion” indices does not predict lead release.

A report by Gregory (1985) entitled “Galvanic Corrosion of Lead in Copper Pipework: Phase II, Effects of Treatment on Galvanic Corrosion Potentials” is often cited as one of the original contributions to the CSMR index. Under laboratory conditions, Gregory (1985) used corrosion cells with lead and copper coupons that were connected with an external circuit. The corrosion cells were used to study four water sources that had CSMR ranging from 0.37 to 1.35. Gregory (1985) found that the Water 3 (CSMR of 1.35; Alkalinity 10 mg/L as  $\text{CaCO}_3$ ; pH 8.3) the addition of sulphate was the least effective treatment resulting in the lowest reduction of corrosion potential. Further, the addition of sulphate was essentially ineffective in reducing corrosion potential in Waters 1 (CSMR of 0.73; Alkalinity 264 mg/L as  $\text{CaCO}_3$ ; pH 7.4), 2 (CSMR of 0.37; Alkalinity 185 mg/L as  $\text{CaCO}_3$ ; pH 7.6), and 4 (CSMR of 0.52; Alkalinity 116 mg/L as  $\text{CaCO}_3$ ; pH 8.5). It should be noted that for this, Gregory (1985) designed his experiments to understand galvanic corrosion between lead solder and copper pipe.

Similarly, work by Edwards and others (e.g., Nguyen et al., 2010; Edwards and Triantafyllidou, 2007) designed experiments to understand CSMR using lead solder and copper pipes. Thus, it is considered an index that may predict the effects of galvanic corrosion between copper and lead pipe and more specifically between copper and lead solder joints. Further, the concept that CSMR only pertains to galvanic couples is emphasized by Arnold, et al. (2020), based



on a survey of 60 utilities, with the important comment that “[a]pproximately 75% of the systems in the survey had CSMR values greater than 0.5. No correlations between CSMR values and 90th percentile lead values were observed when evaluating the systems in this study.” Consistent with this literature, Dr. Edwards’ acknowledged that CSMR “isn’t completely understood” during his deposition.

Further, due to the complexities and the factors that contribute to lead release, the use of such indices like CSMR for corrosion control is not practical and has a very limited scope in Flint. Based on analysis of lead-bearing pipe scales conducted during a USEPA pipe rig study conducted by Williams et al. (2018), it is evident that such solubility models or corrosion indices may not be appropriate. As described by Williams et al. (2018) in a discussion regarding lead scale on pipe extracted from Flint:

Pipe scale analysis revealed the complex nature of the lead pipe scales in Flint, which contain a significant amount of amorphous material, making a priori treatment effectiveness computations and predictions essentially impossible.

In other words, the complex properties of the scale in the Flint system, as characterized by the USEPA, make it nearly impossible to predict lead release using solubility models and indices. Thus, using such metrics would not have provided reliable information on controlling lead release. From the context of practice, along with basic water quality information such as pH, these indices would help to provide a very high level understanding of corrosion control requirements. It is expected that the amorphous scale impacts would have been heightened when elevated NOM and iron concentrations were present in the Flint system in 2014.

### **3.2 Lead Release and Natural Organic Matter**

Natural organic matter (NOM) is present in virtually all drinking water sources, but its concentration and chemical properties vary from source to source. The origins of NOM also vary and may include sediment, surface runoff, and aquatic or terrestrial plants and/or animals. Microorganisms in surface water sources can also produce NOM. The origins of NOM affect its chemical properties and its removal during drinking water treatment. For example, natural organic matter generated through the decomposition of plant material is generally humic in nature and is known to have hydrophobic (dislikes water) properties. On the other hand, microbially produced (biogenic) natural organic matter tends to be hydrophilic (water-loving).

Exposure to NOM in drinking water is very common and is not associated with direct health effects. However, the presence and characteristics of NOM have significant impacts on drinking water treatment processes aimed at protecting public health. Health Canada (2019) noted that NOM can contribute indirectly to health impacts in drinking water through the following pathways:

- a deterioration of pathogen removal capability due to increased coagulant demand, which can lead to suboptimal coagulation conditions;

- a deterioration of pathogen inactivation capability due to chemical disinfectant demand or interference with ultraviolet (UV) disinfection;
- the formation of disinfection by-products (DBPs);
- the development of distribution system biofilms that can harbor pathogens; and
- potential increases in lead and/or copper concentrations as a result of corrosion of lead-and/or copper-bearing materials (e.g., piping, fittings).

NOM is quantified using several measurement techniques. A summary of these measurements is provided in Table 2 for reference purposes. To ensure accuracy/clarity in description throughout the report, working definitions have been provided, and where appropriate, references to regulatory or working knowledge of the parameter are also provided below in Table 2.

**Table 2.** *Various NOM measurement parameters and reference values.*

<b>NOM Measurement Parameter</b>	<b>Description of Measurement</b>	<b>Regulatory / Practice Based Reference Value</b>
<i>Total organic carbon (TOC)</i>	A measure of the total amount of carbon in organic compounds present in a water sample.	TOC removal is described in R 325.10610c <b>Control of disinfection by-product (DBP) precursors; treatment technique of Michigan Safe Drinking Water Rules (see below)</b> <sup>1</sup>  Treated water TOC < 2mg/L in parts of Europe. <sup>2</sup>
Dissolved organic carbon (DOC)	A measure of the “dissolved” amount of carbon in organic compounds present in the filtrate of a 0.45 µm water sample. For filtered water TOC and DOC become similar.	See TOC (when organic matter is mostly dissolved TOC ~ DOC).
UV absorbance at 254 nanometers (UV254)	A measure of the UV light (254 nm) absorbed by organic compounds in a 0.45 µm filtered water sample. This metric conveys information about the concentration and	See SUVA.



	structure of natural organic matter.	
Specific UV absorbance at 254 nanometers (SUVA)	Determined by dividing the UV-254 measurement by the concentration of DOC Absorbance of UV light per unit of dissolved organic carbon.	Generally accepted relationship between raw water SUVA and TOC removal during coagulation (see Table 3).
Chemical oxygen demand	A measure of the oxidizable organic matter in water samples. Correlated with TOC and DOC. A measurement of COD has been estimated to be approximately twice the concentration of DOC for finished water in conventional plants. <sup>3</sup>	European Guideline requirement of < 5mg O <sub>2</sub> /L. <sup>2</sup>

1- MDEQ, 2015. Michigan Department of Environmental Quality (MDEQ), Drinking Water and Municipal Assistance Division Supplying Water to the Public. R 325.10610c Control of disinfection byproduct (DBP) precursors;

2- Government of France, 2007.

3- Stoddart and Gagnon. (2014).

SUVA has been widely used as an indicator of NOM characterization (e.g. hydrophobic vs. hydrophilic) as well as an indicator of coagulation effectiveness for NOM removal in the water industry. Table 3 depicts the generally accepted relationships between NOM, SUVA, TOC and coagulation.

**Table 3.** *Relationship between SUVA, NOM origin, and potential TOC removal via coagulation (Adapted from Health Canada, 2019).*

SUVA (L/mg-m)	NOM Composition	UV Absorbance	Coagulation	Potential TOC Removal
<2	Mostly hydrophilic and low molecular weight compounds (e.g. biogenic in nature – microbially produced)	Low	NOM has little influence on coagulant dose (i.e., NOM recalcitrant to coagulation)	0-40%; higher end for waters with high TOC

2-4	Mixture of hydrophilic and hydrophobic compounds; mixture of molecular weights (e.g. mixture of humic and biogenic)	Medium	NOM influences coagulant dose	40-50%; higher end for waters with high TOC
>4	Mostly hydrophobic and high molecular weight compounds (e.g. humic in nature, produced by decaying plant material)	High	NOM controls coagulant dose	60-80%; higher end for waters with high TOC

### 3.2.1 Treatment Rules for NOM in Michigan Require Enhanced Coagulation to Minimize Total Trihalomethane Concentrations

In Michigan, the Department of Environmental Quality requires, under administrative rule R325.10610c (control of disinfection byproduct precursors; treatment technique), the removal of TOC by surface water facilities (using conventional or lime softening water treatment) with TOC levels above 2 mg/L in their source water. The requirements set out by the MDEQ follow a 2-step process; if Step 1 requirements are not met then the utility must proceed with Step 2 requirements. These treatment steps regulated by the MDEQ are described herein:

**Step 1.** The MDEQ requires specific percentage of source water TOC must be removed to limit disinfection byproduct formation (via enhanced coagulation or enhanced softening). This percentage is a function of the source water TOC and alkalinity (Table 4). If a utility cannot meet this criterion, they can achieve TOC removal through enhanced coagulation as described under Step 2.

**Table 4.** Step 1 MDEQ requirements for TOC removal compliance via enhanced coagulation.  
Source water alkalinity (mg/L as CaCO<sub>3</sub>)

Source water TOC (mg/L)	0-60	>60-120	>120
>2.0-4.0	35%	25%	15%
>4.0-8.0	45%	35%	25%
>8.0	50%	40%	30%

**Step 2.** If Step 1 cannot be met, the utility must evaluate coagulant dosage based on the point of diminishing returns concept: the point at which an additional 10 mg/L of alum fails to remove 0.3 mg/L of TOC (this relationship is defined equivalently for other coagulants). Water systems with SUVA less than 2 L /mg.m or TOC less than 2 mg /L are exempt from the Step 2 enhanced coagulation requirements under the MDEQ.

The concentration and quality of NOM have an influence on the design and operation of water treatment processes. The type and level of treatment should consider the quantity of natural organic matter as well as its source specific nature and short-term variability. Source-specific treatability studies should be conducted to assess NOM removal options and should include bench-scale or pilot-scale testing (Plourde-Lescelleur et al., 2015; Kastl et al., 2016).

A combination of coagulation/flocculation, sedimentation (or clarification) followed by filtration and disinfection with chlorine is a commonly used, industry-wide treatment approach for NOM removal. This conventional treatment process is often accompanied by additional treatment stages like granular activated carbon (GAC) to provide additional removal of NOM.

Small particles in surface water, including natural organic matter, typically exhibit a negative surface charge. Enhanced coagulation is a strategy for removing TOC and controlling the formation of trihalomethanes (THMs) and haloacetic acids (HAAs) in drinking water systems (Edzwald and Tobiasson, 1999). Coagulation is the process of adding a positively charged chemical coagulant to neutralize or destabilize the negatively charged natural organic matter molecules. There are several types of coagulants commonly used in the water treatment industry, including aluminum sulfate (alum), polyaluminum chloride (PACl), aluminum chlorohydrate (ACH), ferric chloride ( $\text{FeCl}_3$ ), and ferric sulfate ( $\text{Fe}_2(\text{SO}_4)_3$ ) (Crittenden, 2012). The coagulated NOM particles form aggregates which collide and adhere with each other, and eventually form larger precipitates called “flocs.” This process is referred to as flocculation. Water quality parameters including pH, alkalinity, and temperature dictate the effectiveness of coagulation and flocculation processes. Engineering textbooks on water treatment (e.g. Water and Wastewater Engineering – Design Principles and Practice by Davis, 2010) note that applying coagulation for TOC (enhanced coagulation) removal require higher doses ferric chloride or alum (in the order of 50 to 100 mg/L) and can provide TOC removal of between 30 and 50% (Davis, 2010).

Jar testing is a bench-scale laboratory procedure (Figure 3) that simulates coagulation and flocculation and is used to establish the point of diminishing returns – the optimal coagulant dose to achieve maximum removal of TOC. The basic premise of jar testing is to maximize TOC removal without overdosing coagulant. Water samples are placed in jars with mixers and various amounts of chemical are added to each jar, stirred, and settling of flocs is observed. Jar testing is widely used and is considered standard best practice in the water treatment industry to evaluate and compare various coagulant doses and coagulation conditions (e.g. pH, alkalinity, mixing speeds) in terms of TOC removal. Furthermore, jar testing can be used to identify point of diminishing returns (PODR) doses. The MDEQ PODR states incremental addition of 10 mg/L of alum or ferric coagulant result in a TOC removal of less than or equal to 0.3 mg/L. Jar tests can be used to ensure compliance with this rule. For example, alum would be dosed incrementally by 10 mg/L in a jar test, and the point where the addition of 10 mg/L of alum does not yield at least 0.3 mg/L reduction in TOC would be considered the PODR. The PODR dosage for the city of Flint

was determined to be 90 mg/L of ferric chloride through jar testing by Veolia North America (“VNA”).



**Figure 3.** *Standard jar test apparatus.*

NOM reacts with chlorine to form potentially toxic disinfection by-products such as THMs and HAAs. NOM also consumes free chlorine residual during the process of disinfection byproduct formation. These are regulated parameters under the EPA’s Disinfectants and Disinfection Byproducts Rule. The regulated concentrations for THMs and HAAs are 80 and 60 µg/L, respectively. The compliance for trihalomethanes and haloacetic acids is determined on a Local Running Annual Average (LRAA) calculation of samples collected from the last four quarters.

When the concentration of NOM is high, the applied chlorine dosages must also be high to overcome the chlorine demand exhibited by the organic matter and to maintain a free chlorine residual, which results in excessive formation of THMs. Thus, it is critical to optimize the removal of NOM through treatment such as enhanced coagulation.

In 2014, Flint’s drinking water had exceedingly high levels of NOM, as indicated by THM levels measured in the distribution system. A 2014 survey of THMs in 394 larger (>100,000 customers) public water systems in the United States found upper and lower quartile THM concentrations of approximately 43 and 17 µg/L, respectively (Seidel et al. 2017). In 2014, the disinfection byproduct class of THMs exceeded their regulated values in the Flint system. Goodwill et al. (2018) noted that the *lowest* measured THM concentration in the system during August 2014 was above the locational LRAA regulatory limit of 80 µg/L. Additionally, the maximum result approached 200 µg/L, and the median was 134 µg/L. According to the survey conducted by Seidel et al. (2017), the Flint system was in the top fifth percentile for highest THM concentration in the United States (for cities greater than 100,000 customers). In other words, Flint was among 20 water utilities in the United States with the highest THM concentrations and the maximum value of 200 µg/L would be among the highest reported by Seidel et al. (2017).

Such high THMs are indicative of poor removal of organic natural matter. Chemical disinfectants such as chlorine react with NOM to form potentially harmful disinfection byproducts such as THMs and HAAs. Per the MDEQ, routine monitoring for THMs and HAAs must be conducted every 3<sup>rd</sup> month and values must be below 80 and 60 ug/L, respectively.

On May 22, 2014, for example, the TOC concentration of the Flint River was 10.3 mg/L (Masten et al., 2016), while one day earlier (on May 21<sup>st</sup>), the THM concentrations in the distribution system were 162.4 ug/L (Table 5). In light of this connection, it was clear that the plant was challenged by the NOM in the Flint River and, as a result, THM concentrations exceeded regulatory values. As described by Masten et al. (2016), total THM concentrations at several sampling locations exceeded the 80 µg/L regulatory limit during May and August 2014 (Table 5).

**Table 5.** *Trihalomethane concentrations in the Flint distribution system in Summer 2014 (Adapted from Masten et al., 2016).*

Location	May 21, 2014	August 21, 2014
1	162.4	145.3
2	75.1	112
3	111.6	127.2
4	79.2	181.3
5	106.4	196.2
6	82.2	112.4
7	88.2	144.4
8	96.5	118.3

Additional THM data for the eight sampling sites were obtained through compliance communication from the MDEQ (MDEQ, September 10, 2014). As shown in Table 6, THM concentrations were reported as an operational evaluation limit (OEL), which is an indicator of operational performance. The OEL is a calculated value based on data since the switch to the Flint River (2x current quarter + previous quarter) divided by 4). The OEL was compared with the State MCL of 80 ug/L to determine the likelihood of exceeding the MCL. In all but one site, the calculated OEL exceeded the State MCL of 80 ug/L.

**Table 6.** *Trihalomethane concentrations and Calculated OEL values for 8 sample sites in the Flint System in 2014.*

Site:	THM (ug/L)		Calculated OEL
	5/21/14	8/21/14	
1	162.4	145.3	113
2	111.6	112	84
3	96.5	127.2	88
4	106.4	181.3	117
5	75.1	196.2	117

6	82.2	112.4	77
7	88.2	144.4	94
8	79.2	118.3	79

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The data shown in Table 6 demonstrates that in 2014, seven out of eight sites exceeded the limit of 80 ug/L for THMs. Further, with respect to the calculated OEL value, six of the sites were well above 80 ug/L, while two of them were more than 95% of the 80 ug/L limit.

The MDEQ required that the City of Flint complete an operational evaluation to minimize future exceedances of TTHM. The MDEQ recommendations for the operational evaluation required the City of Flint to examine treatment operational practices (including treatment changes and treatment problems that contribute to THM formation), as well as examination of distribution operational practices. VNA provided several recommendations to improve treatment for the removal of NOM and the subsequent improvement of THM formation.

Although high NOM levels are not predictive of lead release, their presence can have, under some circumstances, a direct effect on lead in drinking water. If water systems do not remove organic molecules effectively, observational, theoretical, and experimental evidence suggests that they can contribute significantly to lead release. In short, NOM can interact directly with lead by: 1) binding with lead, increasing its solubility, 2) dispersing lead colloids, leading to increased mobility, and 3) reducing lead to more soluble forms.

An experiment by Colling et al. (1992) used a laboratory scale lead pipe rig to evaluate plumbosolvency (the ability of water to dissolve lead) of hard waters in the presence of low concentrations of organic rich water such as humic substances that is typical of peat or other similar earthen materials. Experiments were performed with the addition of humic material to waters with low plumbosolvency. Peat water (organic rich water) was added between 10 and 1000 mL to 25 L water with low plumbosolvency. Actual TOC concentrations ranged from 0.6 to 1.7 mg/L TOC (Colling et al., 1992). Table 7 below shows the results from this experiment.

**Table 7.** *Effect of additions of humic matter on low plumbosolvency waters (from Colling et al. (1992)).*

<b>Addition of peat water to low plumbosolvency water (25 L)</b>	<b>Plumbosolvency result (ug Pb/L)</b>
No addition	30
+ 10 mL peat water	26
+ 50 mL peat water	51
+ 75 mL peat water	82
+ 100 mL peat water	97
+ 250 mL peat water	114
+ 1000 mL peat water	102

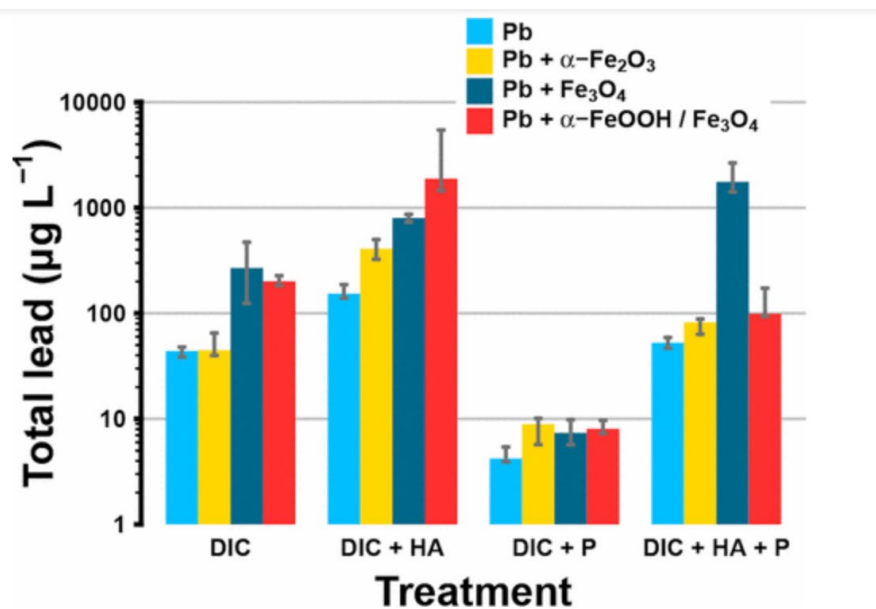
As depicted in Table 7, the addition of humic material (e.g. peat water) to low plumbosolvency water increased the plumbosolvency (lead release). Increasing the humic content corresponded with higher lead concentrations in waters that were previously considered to have low plumbosolvency. This study also surveyed treated waters and found that waters with the highest TOC concentration ( $>3$  mg/L TOC) were more plumbosolvent, while waters with low TOC ( $<0.6$  mg/L) were less plumbosolvent. For reference, treated water in Flint ranged from 3 to 6 mg/L (see Table 8). The recommendation from VNA would be 90% removal of NOM to achieve a DOC of approximately 0.7 to 1 mg/L (based on raw water values of 7 to 10 mg/L, Table 8). Such reductions in TOC would have made treated water from Flint less plumbosolvent as described through Colling et al. (1992).

### **3.2.2 NOM's Effects on Lead Mobility**

Further evidence from Colling et al. (1992) demonstrated that water passing through GAC columns provided lower plumbosolvency. Humic addition to hard water ( $\sim 300$  mg/L  $\text{CaCO}_3$ , Anglian Water, 2018) was the cause of high plumbosolvency. In Flint, the hardness of 340 mg/L  $\text{CaCO}_3$  was comparable to the water studied by Colling et al. (1992). VNA recommended the implementation of GAC filtration to provide additional NOM treatment; this action would have been beneficial for lead control.

Even in the presence of corrosion inhibitors, the presence of NOM can promote lead release. Corrosion inhibitors such as orthophosphate can decrease the dissolution of scales (e.g. hydrocerussite and cerussite). Orthophosphate is often added to inhibit corrosion, and it has been shown to mitigate lead release. Trueman et al. (2017) evaluated the impact of upstream iron corrosion on lead mobility in water distribution systems. Humic acid, dissolved inorganic carbon, and orthophosphate were variables considered in the experimental design, and lead was coupled with various iron minerals to simulate galvanic corrosion. Figure 4 below summarizes results from Trueman et al. (2017).





**Figure 4.** Lead release as a function of humic acid, dissolved inorganic carbon, and orthophosphate (from Trueman et al., 2017).

Trueman et al. (2017) found that the presence of humic substances increased lead mobility due to both uniform and galvanic corrosion compared to the treatments without humic substances. Further, the total lead released was highest in the systems where lead was coupled to iron minerals and where humic acid was present (e.g. DIC + HA treatment). The addition of orthophosphate lowered the lead release, but the addition of humic substances made the addition of orthophosphate less effective. Overall, results of this work suggested that the addition of humic substances increased lead release by more than 4-fold, and that orthophosphate was less effective in the presence of humic material (Trueman et al., 2017). It was also noted that in water systems that do not have optimized removal of NOM, lead contamination events were driven by elevated concentrations of NOM and that optimized coagulation or adsorption processes would minimize the effect of organic matter on lead mobilization (Trueman et al., 2017).

Zhou et al. (2015) found that increasing NOM (from 1 to 7 mg/L DOC) resulted in significant lead release in an experiment with simulated partial lead service line replacements. Trueman et al. (2019a) found that natural organic matter as well as iron and manganese influence transport of lead from source to tap; in tap water lead was stabilized by natural organic matter. In another study, Trueman et al. (2019b) found that the presence of humic substances provide additional capacity of distributed water to transport lead and that their removal through treatment could reduce lead exposure. Zhao et al. (2018) noted that the removal of organic carbon during water treatment to lower the formation of disinfection-byproducts may have the additional benefit of minimizing the mobility of lead-containing particles.

Korshin et al. (2005) found that natural organic matter prevented the formation of lead scales such as cerussite, altered the formation of hydrocerussite, and thus the concentrations of soluble lead were greater in the presence of DOC.



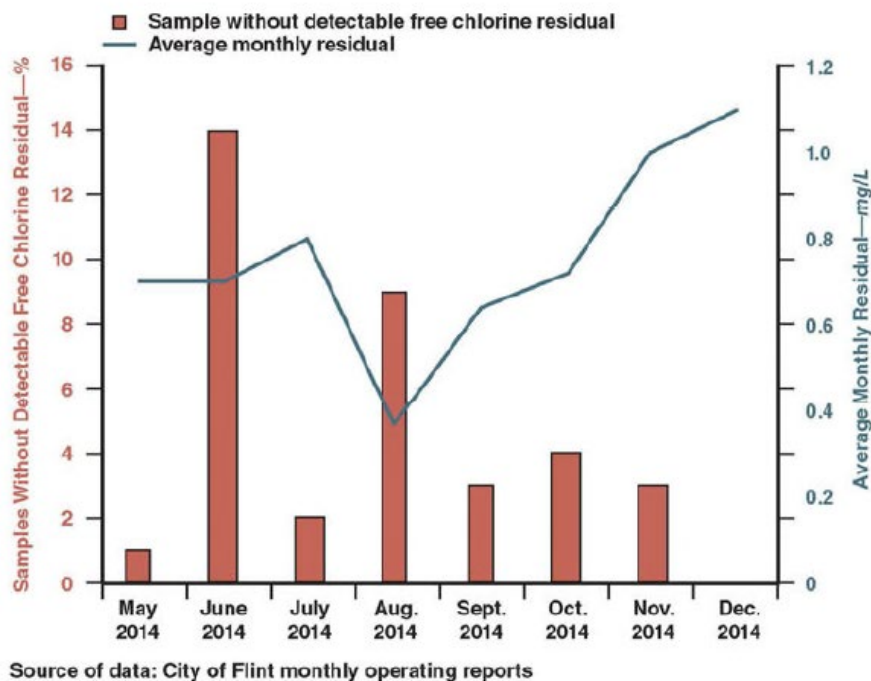
Korshin et al. (1999) found that natural organic matter increased both short- and long-term leaching of lead, and was most noticeable at concentrations of approximately 2 mg/L of DOC (Korshin et al., 1999). A study by Korshin et al. (2019) evaluated the dispersion of lead colloids from scale in the presence of natural organic matter. Korshin et al. (2019) demonstrated that in the presence of NOM, there was a shift in the surface charge of the scale resulting in the dispersion of lead colloids, and that the addition of phosphate reduced the release of lead colloids.

Through its interactions with chlorine in the distribution system, NOM can also have indirect effects on lead in drinking water. As a consequence of high NOM, Flint's drinking water was characterized by low free chlorine residuals in the distribution system. In drinking water, free chlorine is a measurement that represents the combined concentration of hypochlorous acid and the hypochlorite ion. In general, hypochlorous acid has more disinfection capacity than hypochlorite ion, and at pH levels of 7.6 or less, hypochlorous acid represents at least 50% (or more) of the free chlorine concentration.

### **3.2.3 NOM's Effects on Scale Stability**

"Chlorine residual" is a term that is used to represent the free chlorine concentration available for disinfection in the distribution system. The maintenance of chlorine (or disinfectant) residual has been conducted on the premise that maintaining an adequate disinfectant residual in the distribution system can minimize regrowth of indicator or pathogenic microorganisms, detect microbial intrusion more rapidly than microbial monitoring alone, and reduce the risk of general contamination as a result of a distribution system breach (Haas, 1999). Under the Surface Water Treatment Rule, a detectable disinfectant (e.g. free chlorine) residual of 0.2 mg/L is required at the entrance point to the distribution system. Further, the disinfectant residual cannot be undetectable in more than 5% of the samples each month, for any two consecutive months.

Chlorine demand refers to the reaction of free chlorine (i.e. hypochlorous acid or hypochlorite ion) with dissolved substances such as NOM, iron, manganese and ammonia. Given the water quality in Flint, high concentrations of NOM and metals like iron would have depleted the free chlorine residual in the distribution system. The reaction between NOM and chlorine would yield substantial concentrations of disinfection by products like THMs, resulting in depletion of free chlorine. This is shown in Figure 5 below, which depicts average monthly free chlorine residuals as well as the number of samples without detectable free chlorine residual.



**Figure 5.** Average monthly free chlorine residual and samples without detectable free chlorine residual in the Flint System in 2014 (Masten et al., 2016).

In Figure 5, it can be observed that during the period between May to December 2014, the free chlorine residual in the Flint system was highly inconsistent. With regards to the Surface Water Treatment Rule, in 2014 there were two months (June and August 2014) when more than 5% of samples did not have a detectable disinfectant residual. Further, the lowest average monthly residual (August 2014), when nearly 10% of samples did not have a detectable residual, corresponded with some of the highest THM concentrations reported by the MDEQ in August 2014. In this case, the chlorine residual was most likely consumed by the elevated concentrations of NOM (as well as other minerals like iron and manganese) present in the Flint system. VNA provided several recommendations to reduce NOM concentration and disinfection byproducts.

When chlorine is added to water, it produces hypochlorous acid or the hypochlorite ion, depending on the pH of the water. Hypochlorous is a strong oxidizing agent used in drinking water and is the main form of free chlorine below a pH of 7.6. At a pH above 7.6, the hypochlorite ion is dominant, which is a weaker oxidizing agent in comparison to hypochlorous acid.

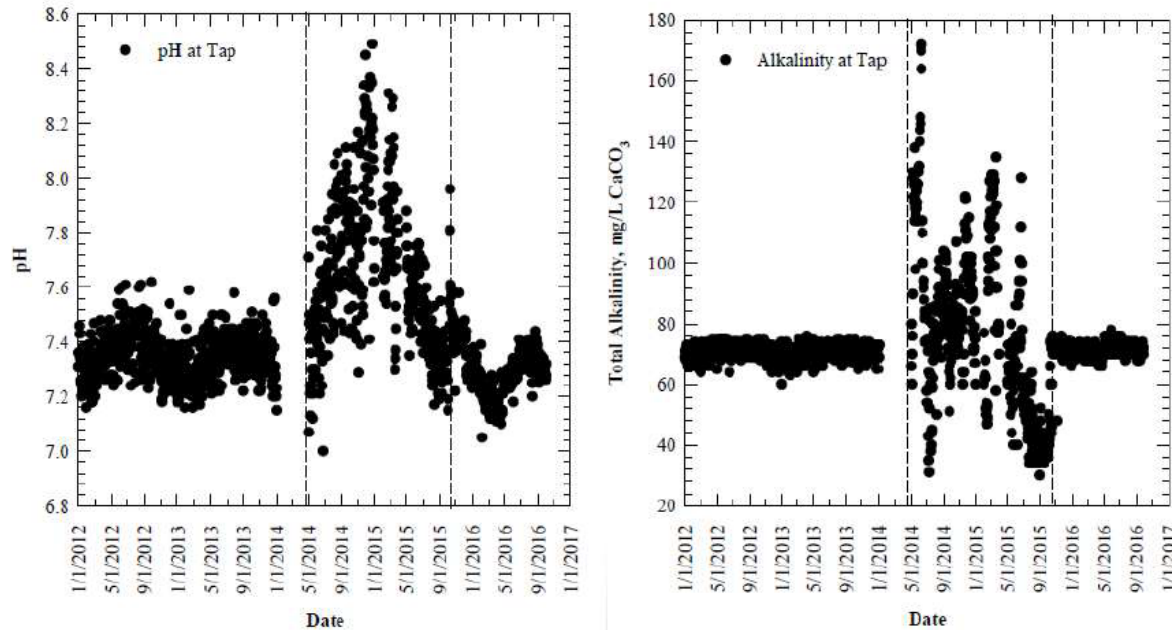
A study was conducted by Lytle and Schock (2005), to understand the stability of lead solids (e.g. scales) under various water quality conditions (e.g. pH and free chlorine residual). In one experiment, free chlorine residual was maintained to obtain a goal concentration of 3 mg/L at a pH of 7.7 to 8.2 for a period of nearly 400 days. During this period, initially the lead mineralogy demonstrated carbonate based lead solids (e.g. cerussite and hydrocerussite), but eventually there was a shift to Pb(IV) dioxide solids. There was a notable change in color when the mineralogy shifted – in the presence of lead carbonate solids, the particles were white, and changed to red/brown when Pb(IV) dioxide was present. Near the end of the study the lead solids reverted

back to carbonate form and the Pb(IV) dioxide was gone. Throughout this period, the soluble lead (e.g. PbII) concentrations changed dramatically and corresponded directly to the changes in lead mineralogy and color. For example, soluble lead concentrations were the highest (0.145 mg/L) during the presence of lead carbonate solids (white), decreased dramatically (to 0.002 mg/L) during the presence of Pb(IV) dioxide (red), and increased again (to 0.113 mg/L) as lead dioxide transformed back to lead carbonate. This suggests that Pb(IV) is stable (e.g. releases less soluble lead).

The cause of the shift in mineralogy was a change in redox potential, which is a direct function of chlorine residual and pH. The chlorine residual in the system was depleted completely during the time of the shift in mineral phase. Lytle and Schock (2005) noted that lead dioxide can form at neutral pH in waters with persistent free chlorine residuals, and that changes in pH, drops in ORP, or both could destabilize these lead dioxide films and thus increase plumbosolvency and possibly increase the release of scale particles. For example, at a pH of 8, chlorine at appropriate dosages are able to achieve the redox potentials sufficient for lead dioxide stability. Overall this suggested that as long as free chlorine was maintained, more stable Pb(IV) dioxide scales formed over time, and a decrease in soluble lead (PbII) was observed (Lytle and Schock, 2005).

The concept of scale stability was further discussed in a presentation on behalf of the USEPA by Triantafyllidou and Schock (May 21, 2019). They noted that “[d]isinfectant demand in distribution systems must be controlled and enough free chlorine consistently maintained through lead service line area to keep protective Pb(IV) corrosion scales in place.”

An analysis of the Flint water quality during 2014 was conducted to gain insight on the properties of scale in the Flint distribution system and how this could have impacted lead solubility. Figure 5 depicts the average monthly chlorine residual in the Flint distribution system during the May to December 2014 (blue line); while Figure 6 shows pH and alkalinity of the treated water (vertical dashed lines represent the period where raw water was supplied from the Flint River).



**Figure 6.** *pH (left) and alkalinity (right) of the treated water supplied in Flint from 2012-2017. The vertical dashed lines represent the period when the system was supplied with Flint River water (May 2014-October 2015).*

As portrayed in Figures 5 and 6, there was extreme variability in chlorine residual, pH and alkalinity during the time when the system was supplied Flint River water during the period May 2014- September 2015. The free chlorine residual started at ~0.7 mg/L in May 2014, dropped to ~0.4 mg/L in August 2014, and increased to ~1.1 mg/L in December 2014.

In comparison, the pH of the treated water varied between approximately 7.0 and 8.5 for the same period, while the alkalinity ranged between 60 and 180 mg/L  $\text{CaCO}_3$  in May 2014, dropped to <40 mg/L  $\text{CaCO}_3$  in August to September and then fluctuated between 60 and 120 mg/L  $\text{CaCO}_3$  between September and December 2014. This is in contrast with a consistent pH of 7.2 to 7.6 mg/L and average alkalinity of ~70 mg/L maintained before May 2014 when the water was supplied by the DWSD.

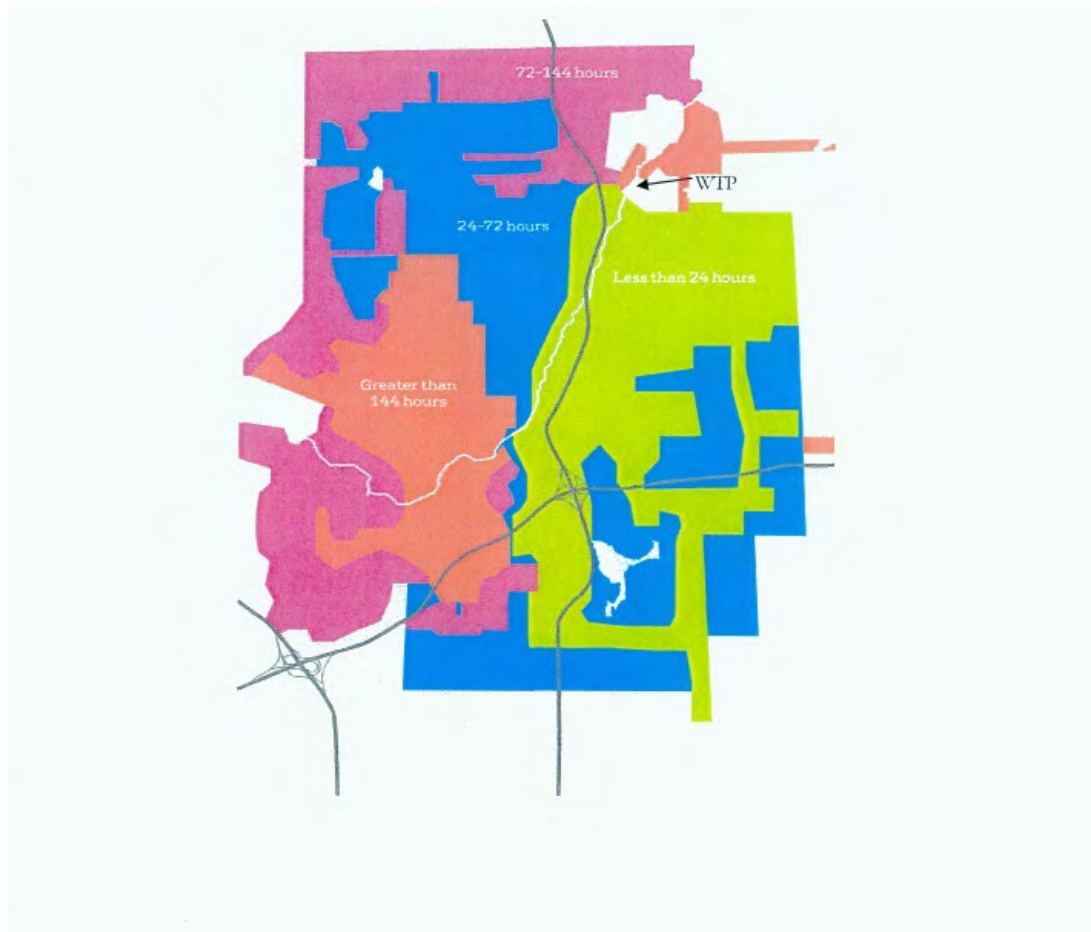
As noted by Masten et al. (2016), in May 2014 the monthly operating records (MORs) noted that the supervisory control and data acquisition (SCADA) upgrades were incomplete; chlorine monitoring equipment had not been installed and the plant had limited polymer available. These items noted in Masten et al. (2016) highlighted just a few plant deficiencies in 2014, which would have resulted in the observed variability in pH, and alkalinity. As noted by Edwards (2020) at his deposition “pH is the most important factor in lead corrosion if you are not adding phosphate.”

In addition to direct impact on lead stability, pH impacts the speciation of chlorine (e.g. weak oxidant hypochlorite ion vs. strong oxidant hypochlorous acid). For instance, starting in summer 2014 the pH was almost always above 7.6 (approaching 8.5), suggesting that the hypochlorite ion, which is a weak oxidant, would be the dominant form of free chlorine in the Flint

system. In addition, Flint also had occurrences of very low or no chlorine residual during the summer of 2014. The resulting combination of a weaker oxidant with low chlorine residuals would magnified have lead instability.

It should be noted that, independent of NOM, water age impacts chlorine residuals in the distribution system. Water age is a term that is often used to describe the amount of time water remains in a distribution system. In general, water age is a predictor of decreasing water quality, such as increased THMs and lower chlorine residuals. As water ages in a distribution system, chlorine residuals are more difficult to maintain as chlorine begins to react with other substances in the water. For example, chlorine (as hypochlorous acid/hypochlorite ion) is highly reactive with iron and chlorine will oxidize iron pipe over time.

The significance of water age was raised in the March 30, 2015 Technical Memo by VNA. In particular, it was noted that “water age appears to be greatest on the west side of the system, corresponding to areas of the highest THM concentrations. The water age map [see Figure 7] also illustrates operational difficulties in the system possibly due to closed or broken valves.”



**Figure 7.** “Predicted Water Age Map of Flint, Michigan Based on Existing Hydraulic Model” as reported in the March 30, 2015 Technical Memo by VNA

Friedman (2017) reported that main breaks for 2015 and 2016 were 263 and 129, respectively. In general, water main breaks will increase water age. Water main breaks will also contribute to reduced water quality, such as high iron concentrations and reduced chlorine residual.

As noted in the City of Flint (2017) Final RTRC Coliform Monitoring Plan, Sampling Site 1 has address of 2501 Flushing Rd - University Market. Comparing the City of Flint (2017) Final RTRC Monitoring Plan with Figure 7, Sampling Site 1 would likely have a water age of approximately 72-144 hours (or longer). Friedman (2017) presented chlorine and total iron concentrations for sampling site 1 for the period of November 2015 to September 2016. For Site 1, it was found that in November to December 2015 iron concentrations ranged from 0.08-mg/L to 0.16 mg/L and chlorine concentrations were less than 0.2-mg/L. For Site 1 in August 2016 to September 2016, chlorine levels exceed 1.0-mg/L and iron concentrations were less than 0.02-mg/L.

The data presented by Friedman (2017) align with water age and chlorine residual recommendations presented in the March 30, 2015 Technical Memo by VNA. In that report, VNA recommended:

- “Maintain sufficient chlorine feed after filtration to meet CT requirements and system chlorine residuals.”
- “Flushing mains, eliminating storage tanks, replacing broken valves etc. all are expected to help reduce water in the system and to improve water quality.”
- “Consider chlorine decay testing and development of a chlorine decay model. Decay models can help define flushing durations to minimize water age and to improve distribution system water quality.”

### **3.3 VNA’s Recommendations Relating to NOM**

As part of its work in Flint, VNA provided several treatment and operational recommendations surrounding the reduction of NOM with subsequent compliance with the EPA disinfectants and disinfection byproduct rule. These recommendations were consistent with the industry best practices described above. The suggested trihalomethane reduction actions made by VNA include: 1) increasing coagulant dose to optimize NOM removal, and 2) installation of granular activated carbon to improve NOM removal.

In its March 30, 2015 Technical Memo, VNA also made several recommendations to address chlorine directly, namely:

- “Increase process control monitoring to include each unit process in the treatment plant and any necessary water quality parameter to make more informed decisions related to chemical treatment and more consistent control for chemical dosing.”
- “Eliminate midpoint chlorination following acceptance by MDEQ of the CT calculations using only post filtration chlorination in plant operations. This might require installation of a chlorine feed point in the common filter effluent well just downstream of the filters.



Maintain sufficient chlorine feed after filtration to meet CT requirements and distribution system chlorine residuals.”

- “Consider chlorine decay testing and development of a chlorine decay mode. Decay models can help define flushing durations to minimize water age and to improve distribution system water quality.”

Implementing these recommendations would have allowed for the maintenance of consistent free chlorine residual and would have contributed to the development of a stronger scale, as “distribution systems that have historically maintained high chlorine residuals could have PbO<sub>2</sub> functioning to limit lead release.” (Lytle and Schock, 2005).

Substantial reductions in NOM through the Flint treatment process would have been necessary to reduce the THM concentrations described previously, and to achieve sufficient TOC removal. In Michigan, the MDEQ requires removal of TOC by surface water facilities using conventional or lime softening water treatment with TOC levels above 2 mg/L in their source water. The requirements set out by the MDEQ follow a 2-step process; if Step 1 requirements are not met, then the utility must proceed with Step 2 requirements. Briefly, Step 1 requires a specific percentage of source water TOC to be removed via enhanced coagulation. This percentage is a function of the source water TOC and alkalinity (Table 4). If a utility cannot meet this criterion, they can achieve TOC removal through enhanced coagulation as described under Step 2. Under Step 2, the utility must evaluate coagulant dosage based on the *point of diminishing returns concept*: the point at which an additional 10 mg/L of alum fails to remove 0.3 mg/L of TOC.

Table 8 shows TOC concentrations in the Flint River and Flint treated water during 2014-2015 from different available reports. For comparison, water quality from the Detroit Water Supply District was also included in Table 8.

**Table 8.** *Comparison of TOC in the Flint River, Flint treated water, and treated water from the Detroit Water Supply District.*

Parameter	Flint River	Flint Treated Water	Detroit Water Supply District (2015-2017)
TOC (mg/L)	7.05 <sup>1</sup> – 10.3 <sup>2</sup>	3.0 <sup>2</sup> , 5.4 - 6.1 <sup>3</sup>	~ 1.5 - 1.9 mg/L (est. based on COD) <sup>4</sup>

1-VNA Technical Memo March 30, 2015;

2- Masten et al. (2016);

3 –HOH Water Technology (Lab report water analysis Flint system, January 2015);

4 – Estimated based on calculation TOC or DOC = ½\*COD from Stoddart and Gagnon (2014);

Under the MDEQ enhanced coagulation requirements, the City of Flint would have been required to remove 30% of TOC to meet Step 1 requirements. Although Step 1 requirements may have been met, there remained substantial concentrations of TOC in the treated water. Flint River water had a nominal TOC of 10-mg/L (Goodwill et al., 2018), and a SUVA of 2.8 L/mg.m. Thus, the Flint treatment plant was not be exempt from the enhanced coagulation requirements outlined by the MDEQ. The SUVA value also suggests that the NOM in the Flint River favored removal through enhanced coagulation. However, the treated water TOC concentrations in the Flint distribution system were recorded to be as high as ~6 mg/L. For comparison, the DWSD treated water had concentrations of TOC of less than 2 mg/L. Others (e.g., Masten et al., 2016) have also pointed to necessary reductions in NOM to achieve compliance with the MDEQ TOC removal requirements. Further, this was confirmed by the MDEQ, which stated that:

The water system must take proactive steps to be able to comply with the MCL's for TTHM and HAA5. While it is recognized that the removal of Total Organic Carbon disinfection by product precursors as part of the treatment process has performed above minimum removal requirements under the treatment technique, around 4 parts per million of organic carbon are still present in the treated water and may be leading to additional TTHM formation in the City's distribution system. (MDEQ, September 10, 2014)

The data described here highlight inadequate TOC removal at the Flint plant during 2014, which was further evidenced by elevated THMs in treated water. As described above, substantial reductions in NOM through the Flint treatment process would have been necessary to achieve sufficient TOC removal to minimize DBP formation, despite having achieving Step 1 removal requirements (e.g. 30%). This would signal further evaluation of coagulant addition through the Step 2 requirements, which are achieved through the point of diminishing returns concept, described previously.

VNA provided a Step 2 point of diminishing returns (PODR) analysis (March 30, 2015 Technical Memo) and noted that “the point of diminishing return for FeCl<sub>3</sub> appears to be at about 90 mg/L.” Further the VNA March 12, 2015 Water Quality Report directed the Flint plant to “increase the ferric chloride dosage to 100-mg/L depending on river water TOC levels.” In both circumstances, VNA demonstrated a need to be compliant with MDEQ coagulation conditions for TOC removal. Weighted against the benefit of improved NOM control, an increased ferric chloride dosage would not have had any significant effect on lead release.

It is important that jar testing and DBP formation potential testing be performed to determine the feasibility of optimized coagulation for NOM removal. VNA recommended that the City of Flint increase coagulant dosing to optimize TOC removal and to control subsequent formation of THMs. VNA conducted coagulant screening through jar testing to optimize the coagulant dose and to test alternative coagulants with a goal of maximizing TOC removal and minimizing THM formation.

VNA conducted jar testing to evaluate several coagulants including ferric chloride, ferric sulfate, polyaluminum chloride (PACl), and aluminum chlorohydrate (ACH) for use the Flint plant. These are industry accepted coagulants used to improve coagulation and removal of NOM



to minimize THM formation (Crittenden et al, 2012). Jar testing conditions simulated the current treatment process at the Flint plant (e.g. detention times and mixing conditions). At the time of the study, the current ferric chloride dose at the Flint plant was approximately 40 mg/L. Through jar testing, VNA evaluated coagulant doses from 10 to 100 mg/L in increments of 10 mg/L to establish the point of diminishing returns for each coagulant tested.

Results from jar testing conducted by VNA demonstrated that PACl and ferric sulfate were not effective coagulants for Flint water, as TOC removal for these coagulants was less effective compared to the other coagulants tested. Ferric chloride and ACH provided up to 40 and 50% removal of TOC, respectively. However, given the fact that ACH costs twice as much as ferric, they recommended that Flint continue coagulating with ferric chloride, which was already approved for use by the MDEQ. Further jar testing with ferric chloride demonstrated a point of diminishing returns dosage of 90 mg/L, which was over twice as much as what the Flint plant was dosing at the time.

VNA recommended ferric chloride dosages of up to 100 mg/L to improve full-scale TOC removal. This recommendation is consistent with engineering textbooks on water treatment (e.g. Water and Wastewater Engineering – Design Principles and Practice by Davis, 2010) which notes that applying enhanced coagulation for TOC removal require higher doses ferric chloride or alum (in the order of 50 to 100 mg/L) and can provide TOC removal of between 30 and 50%. The SUVA value for the Flint River (2.8 L/mg.m) suggests that up to 50% TOC removal could be achieved through enhanced coagulation, and this value was corroborated through jar testing by VNA.

The dose recommended by VNA would allow Flint to exceed NOM removal percentages required by the MDEQ (Table 5) under Step 1 and Step 2 enhanced coagulation requirements. For example, a TOC of ~7 mg/L and total alkalinity of >230 mg/L as CaCO<sub>3</sub> for the Flint River (obtained from VNA Technical Memo, March 30, 2015) would result in a requirement of 30% TOC removal via enhanced coagulation. VNA's recommended ferric chloride dose of 90 mg/L based on jar testing and point of diminishing returns would have achieved at least 40% reduction in TOC (VNA Technical Memo, March 30, 2015). Implementing this increase in ferric chloride dose would have resulted in compliance with MDEQ Step 1 and Step 2 requirements and minimizing of THM concentrations.

The increase in ferric chloride recommended by VNA was a change that could have been implemented immediately by Flint staff to control the removal of TOC. Further, VNA suggested post-clarification TOC values of less than 3.5 mg/L based on jar testing results (compared to raw water values of approximately 10 mg/L). Implementing this strategy would have maximized TOC removal during coagulation and would have resulted in reductions in THM formation during chlorination.

In its analysis and recommendations, VNA noted that in addition to increasing coagulation dosing, additional TOC removal would need to come from granular activated carbon (GAC) to reduce THMs when using the Flint River source. Granular activated carbon (commonly referred to as GAC) can be used to further reduce TOC concentrations in coagulated water. GAC can be used as a filter media that provides a surface on which NOM can adsorb to. GAC has shown to be

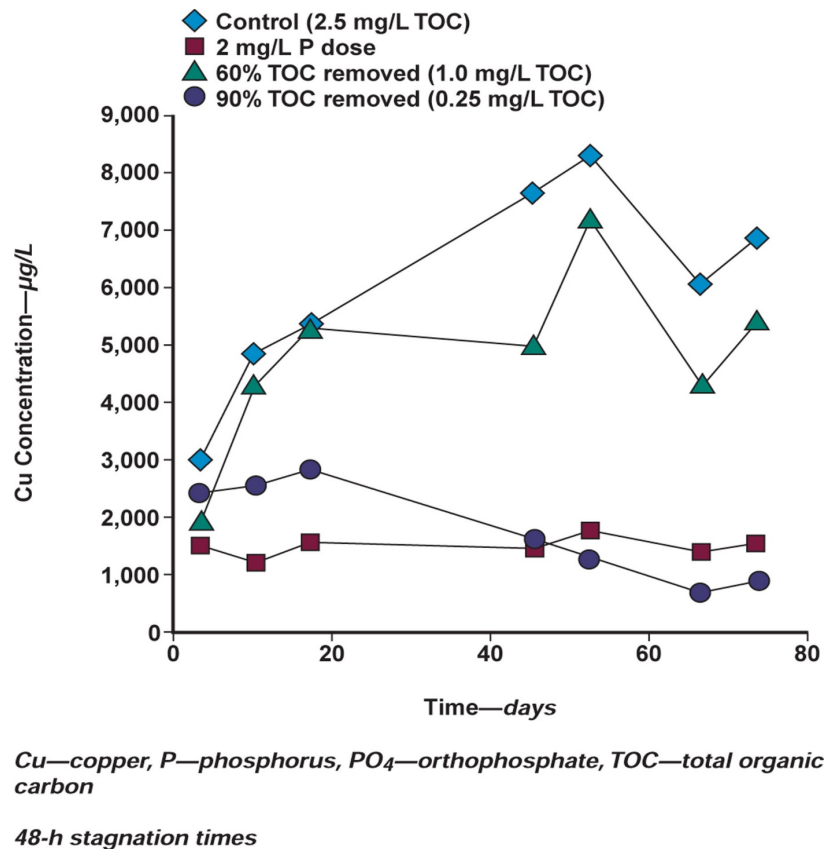
an effective strategy for the removal of NOM and thereby decreasing the amount of THMs formed upon chlorination. The following points were made by VNA concerning TOC removal with GAC:

- “The plant by design is limited on the amount of TOC removal. A maximum removal of only 60% is likely if the plant is properly optimized. The change of filter media to GAC would provide the best reduction possible and provide better than 90% removal dramatically reducing the potential of TTHM formation and thus ensuring compliance with that parameter for the water system.” (March 12, 2015, VNA Water Quality Report).
- “Contract with your engineer and initiate discussions with the State on the reduction of chlorine prior to the filters and changing the filter media to GAC.” (March 12, 2015, VNA Quality Report).
- “Install GAC in the existing filters after the removal of the anthracite layers. A GAC depth of 30-inches is suggested to maximize EBCT [empty bed contact time] to the extent possible and to gain the longest use of GAC for TOC before converting to the lake source water.” (March 30, 2015, VNA Technical Memo).

In addition to process improvements, VNA recommended installation of an online TOC analyzer to monitor raw and treated water TOC concentrations. This would have helped the city of Flint to monitor TOC concentrations to improve long-term operations and control. Furthermore, such analyzers would provide the opportunity to make process adjustments based on fluctuations in TOC concentration.

VNA suggested that the finished water TOC concentration target be set at a value of no greater than 2 mg/L to minimize THM levels in the drinking water. A TOC standard of 2 mg/L would exceed US regulatory standards but would align with international best practices and regulations. The recommendations provided by VNA to increase the ferric chloride dose of 90 mg/L and to implement GAC as a filter media replacement would be sufficient to achieve a treatment goal of 2 mg/L. Per VNA, the change of filter media to GAC would possibly provide better than 90% removal of organic matter, which would yield approximately 1 mg/L TOC in the treated water based on the data for the Flint River in Table 8.

This level of TOC removal would also have a positive impact on copper corrosion, to the extent any was occurring. For example, Arnold et al. (2012) demonstrated that 90% removal of TOC through GAC treatment could achieve stable copper scale formation that would be comparable or even slightly better than the addition of 1-mg/L of orthophosphate (Figure 8).



**Figure 8.** Controlling copper corrosion in new construction by organic matter removal (as reported by Arnold et al. 2012).

### 3.4 VNA's Recommendations on Corrosion Control

Lead release in drinking water can be controlled in the long term by adjusting pH and/or alkalinity, and by introducing corrosion inhibitors. The effectiveness of inhibitors is also dependent on maintaining the distribution system pH and alkalinity. The most commonly used inhibitors include phosphates. Phosphates are used in water treatment as a corrosion control chemical either in the form of orthophosphates for lead corrosion control, polyphosphates to sequester iron, or a blend of the two.

Polyphosphates have been used historically to reduce colored water caused by iron corrosion (e.g. red water). For example, a journal article published in 1957 refers to “glassy phosphates” (sodium phosphate glass) as a water treatment method (dosed in the range of 0.5 to 2 mg/L) to prevent deposition of calcium carbonate scale, and to control iron corrosion, stabilize dissolved iron, or a combination of the two (Illig, 1957). Other studies have reported the use of polyphosphates to prevent iron corrosion (McCauley, 1960; Williams, 1990; Facey and Smith, 1995; Cordonnier, 1997; Maddison and Gagnon, 1999). Polyphosphates sequester iron causing a decrease in red water (Lytle and Snoeyink, 2002). The use of polyphosphates has also been reported to be successful in reducing lead levels (Lee et al., 1989; Hulsman, 1990; Boffardi and

Sherbondy, 1991), however others have reported them as being ineffective at reducing lead (Edwards and McNeill, 2002).

Orthophosphate and zinc orthophosphate are the corrosion inhibitors that are commonly reported as being effective at reducing lead levels in drinking water. Orthophosphate reacts with lead to form compounds that tend to precipitate and form a scale, as opposed to dissolving in water (EPA, 2009). The ability of orthophosphate to control lead release is dependent on the concentration, as well as the pH and alkalinity of the water and the properties of the scale. Typical orthophosphate residuals are recommended to be between 0.5 and 3.0 mg/L (as phosphoric acid) (Vik et al., 1996). According to the EPA (2009), “[c]onventional wisdom is that orthophosphate treatment for controlling lead and copper should target residual concentrations of 0.33 to 1.0 mg/L as P (1.0 to 3.0 mg/L as PO<sub>4</sub>) at the tap when pH is within the range of 7.2 to 7.8. Higher orthophosphate doses (1.0 to 1.2 mg/L as P, or 3 to 3.5 mg/L PO<sub>4</sub> and higher) may be needed.”

Blended phosphates contain a mixture of both orthophosphate and polyphosphate (ratios ranging between 0.05 to 0.7). Blends have been shown to provide both sequestration of metals and reduction in their release (Hill and Cantor, 2011). Per the EPA (2009), “[b]lended phosphate should contain a minimum orthophosphate concentration of 0.5 mg/L as P (1.5 mg/L as PO<sub>4</sub>) as a starting point for evaluation. The orthophosphate ratio in the blend and/or the dose may need to be increased to provide adequate lead control.”

As described in VNA Water Quality Report (March 12, 2015) listed as Priority #2 under the recommended actions it was stated to “[c]ontract with your engineer and initiate discussions with the State on the addition of corrosion control chemical” (page 10). This agreed with the deposition of Dr. Edwards, in which he stated that “the water company hires experts to determine the plan, and the experts determine a plan and that plan is then approved by the primacy agency... the plan includes the optimal water, you know, control parameters, including like minimums of phosphate or pH that have to be observed to achieve a goal of minimum corrosivity.”

Also in Priority #2, the VNA Water Quality Report (March 12, 2015) also stated that “a target dosage of 0.5 mg/L phosphate is suggested for improved corrosion control.” The 0.5 mg/L of phosphate recommended by VNA was in alignment with the EPA Guidance Manual where it is stated that “It is very important for most water systems to maintain a residual of at least 0.5 mg/L orthophosphate as phosphate (P) and if, possible a residual of 1 mg/L as P is preferable.” (EPA, 2003). Considering the finished water quality in Flint at the time, engineers at VNA selected a concentration dose of 0.5 mg/L of phosphate for improved corrosion control, which would have aligned with the EPA Guidance Manual.

With regards to the recommendation of Priority #2 made by VNA, Dr. Edwards also stated that he “would argue it’s a recommendation to begin the conversation with the State. So this is saying, look, you need a corrosion control chemical right. It might not have been in our scope, but, you know, this—you have to begin this discussion...you hire a consultant who knows what they are doing, and they talk about the form and the dose. This is just a starting point. This is not a final recommendation.” Based on my experience, it is routine that a consulting firm may recommend that a corrosion control approach be implemented. However, it is ultimately the utility with the approval of the regulator that would be responsible for implementing the approved corrosion control approach. Despite VNA’s recommendation that (a) the City of Flint initiate discussions

with the State on the addition of corrosion chemical and (b) apply a recommended dose of 0.5 mg/L phosphate, these were not implemented by the City of Flint. The decision on what type of corrosion control chemical (e.g. orthophosphate vs. polyphosphate vs. blended phosphate) would have been discussed during consultation between the engineers and the State if it occurred.

After the switch to the Flint River, there were complaints of red water in the Flint system. Masten et al. (2016) stated “Within a few weeks of the switch, residents started complaining about the color, taste, and odor of their drinking water...During this time, red water and discoloration were observed throughout the distribution system.” Red water comes from iron corrosion which causes the release of particles containing iron in the water. As described previously, Williams et al. (2018) determined that the scale on the lead pipe in the Flint system was amorphous and was enriched with iron, manganese and organic matter. It was also determined that sloughing of this scale resulted in very tiny particles consisting of lead, iron and natural organic matter in the Flint system. Accordingly, the red colored water observed in the Flint system in 2014 was most likely caused by the release of these very tiny particles containing both iron and lead.

Polyphosphates have been used historically to reduce colored water caused by iron corrosion. In a paper by Illig (1957), sodium hexametaphosphate glass (a type of polyphosphate) was described as being used for control of calcium carbonate scale throughout Ohio and other areas of the Midwest, and that over 200 water companies were treating their water with glassy phosphates, including Flint, Michigan (Illig, 1957). It was also noted that although the original application of glassy phosphate was to prevent calcium carbonate scale formation, its most important role was as a corrosion inhibitor (for iron) – “Approximately 1,500 municipalities feed glassy phosphate to prevent “red water” caused by iron pickup in the distribution system” (Illig, 1957). The City of Flint was using polyphosphates over 70 years ago to control red water.

Rahman and Gagnon (2014a) demonstrated that the addition of a blended phosphate (25% zinc phosphate and 75% polyphosphate) would decrease color of iron-based suspensions in water. The observed decreased color present in the blended phosphate system was a result of iron-colloidal particles becoming smaller (<100-nm), electrostatically neutral and increasing in particle density. In a parallel study, Rahman and Gagnon (2014b) demonstrated that a benefit of iron stability was that the addition of this blended phosphate resulted in decreased disinfection by-product formation for three surface water systems.

Although polyphosphates have commonly been used to sequester iron and reduce the occurrences of “red water,” negative aspects of their use have been reported in the scientific literature. For example, Dodrill and Edwards (1995) conducted surveys of US drinking water utilities to evaluate corrosion control on the basis of utility experience and found that lead release was lower for utilities using phosphate inhibitors (e.g. orthophosphates and blended phosphates) and that polyphosphate inhibitors appeared to have an adverse effect on lead release. Trueman et al. (2018) evaluated the use of an advanced analytical technique to demonstrate the effects of orthophosphate and polyphosphates on lead speciation in several drinking water systems. In that study, it was found that polyphosphate appeared to sequester lead in drinking water, whereas orthophosphate resulted in an overall reduction in lead release (Trueman et al., 2018).

Despite concerns in the literature surrounding polyphosphates and potential lead release, they are still being used in the water treatment industry. For example, McNeill and Edwards (2002) evaluated phosphate inhibitor used at US utilities and reported that in a 2001 survey over 20% of participating utilities were still using polyphosphates (compared to ~20% using orthophosphate, ~30% using a zinc orthophosphate, and the remainder were using a blend of poly/ortho/zinc). Arnold et al. (2020) also reported that among 60 utilities in the United States surveyed: 54% used a corrosion inhibitor of which 28% used a blended-phosphate (e.g., mixture of polyphosphate and orthophosphate). Further, Arnold et al. (2020) noted that the 90<sup>th</sup> percentile lead concentrations for utilities using blended phosphates and orthophosphate was minimal.

Collectively these studies demonstrate that:

- Polyphosphates were previously applied in the Flint water system,
- Polyphosphates are largely used to address red water from iron suspensions in the distribution system, and
- Polyphosphate as a chemical additive is common in the water sector, particularly blended with orthophosphate.

In its March 12, 2015 Water Quality Report, VNA noted that although the focus of the work was to comply with THMs, it recommended the “use of corrosion control chemical polyphosphate to minimize colored water from corrosion of unlined cast iron pipe in the Flint distribution system” and that “many people are frustrated and naturally concerned by the discoloration of the water with what primarily appears to be iron from the old unlined cast iron pipes. The water system could add a polyphosphate to the water to minimize the amount of discolored water.” In its March 30, 2015 Technical Memo, it was recommended by VNA to “Install a polyphosphate feed system to apply chemical to the clearwell for sequestration of iron in the distribution system. Polyphosphate feed rates of about 0.5 mg/L typically minimize iron discoloration issues in distribution piping.” With reference to VNA’s recommended polyphosphate addition, Dr. Edwards noted in his deposition that VNA was “saying let’s start the conversation...you could add a polyphosphate, that could minimize the discolored water, you know. It is not going to make it go away...Polyphosphates could do this, phosphates can work wonders, like someone has got to look at this. This is how I read it.” As noted previously, this opinion expressed by Dr. Edwards’ understanding of VNA’s recommendation would be highly consistent with my experience in the water industry.

## CONCLUSIONS

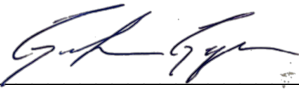
In summary, based on my review of available information, I conclude within a reasonable degree of scientific certainty that:

- Lead service lines were the predominant source of lead in Flint. Through the FAST program, it has become evident that the City of Flint was responsible for and owned the majority of lead pipe in the city.



- The removal of lead pipes in Flint has significantly lowered lead exposure through drinking water in the City of Flint. Research has also pointed to the limited impact that premise plumbing has had on lead concentrations in Flint since 2014.
- Approximately 17% of the homes in Flint had a lead service line on the public or private portion of the address. Within these homes the concentration of lead would be highly dependent on a number of home-specific factors such as the volume of water collected, the temperature of the water, water age, and the length of the service line.
- Lead release in Flint was predominantly from pipe scale that had accumulated on the lead service lines over the long history of the Flint distribution system. The scale contained significant quantities of lead. However, the nature of the scale was amorphous and non-uniform, which resulted in unpredictable release not predictable or describable by theoretical models.
- Indices such as LSI and CSMR are generally not exclusively relied upon for corrosion control in the water industry, and within the context of Flint – which was characterized by amorphous scale on lead pipe surfaces – these indices would in and of themselves have been inconclusive and unreliable for indication of any systemic lead release issues.
- Natural organic matter (NOM) had a significant role in water quality in Flint. High concentrations of NOM resulted in violations of disinfection by-products in Flint. VNA correctly identified that removing NOM would be critical to improving drinking water quality overall and would decrease disinfection by-product concentrations in Flint.
- NOM likely also had a significant role in lead release from scales in Flint. Flint's high concentrations of NOM also would have minimized potential added benefits of corrosion control agents such as orthophosphate. VNA recommended several approaches to minimize NOM which would have improved distribution system water quality including lower lead concentrations. VNA's recommended treatment strategies for NOM removal were consistent with strategies recommended by the State of Michigan as well as those broadly accepted across the drinking water industry.
- VNA's corrosion control recommendations, including consideration of phosphate and/or polyphosphate as a chemical inhibitor additive, were appropriate and supported by scientific knowledge and industry practice.

I provide the foregoing opinions in support of the VNA Defendants' Opposition to the Plaintiffs' Motion for Class Certification. I declare under the penalties of perjury that the statements made are true and accurate and to the best of my information and knowledge.

  
Graham Gagnon, Ph.D., P.Eng.  
Halifax, Nova Scotia

Dated: January 6, 2021.

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**SUMMARY:**

Dr. Gagnon is President of Armview Engineering Ltd., which is a consultancy that is focused on assisting clients in bringing water research into practice. In addition to his role at Armview Engineering, Dr. Gagnon is also a Full Professor in the Department of Civil & Resource Engineering at Dalhousie University, where he also serves as the Associate Vice President of Research for the University. As a Professor, he has held the NSERC / Halifax Water Industrial Research Chair since 2007, which is squarely focused on developing new water research and innovation for the drinking water industry. Dr. Gagnon has authored more than 175 peer reviewed journal articles, 300 conference proceedings and has supervised more than 100 graduate and undergraduate students in his career. Dr. Gagnon is a licensed professional engineer in the province of Nova Scotia, Canada.

In 2014, Dr. Gagnon was awarded the George Fuller Award by AWWA for his commitment and engineering leadership in the drinking water industry. In 2017, Dr. Gagnon joined the Board of Directors for AWWA as nominated through the Atlantic Canada section. In 2020, Dr. Gagnon was named a Fellow of the Canadian Society for Civil Engineering (CSCE) in recognition of his distinguishing contributions to Civil Engineering.

**ACADEMIC BACKGROUND:**

1997	PhD, Civil Engineering	University of Waterloo, Waterloo, ON
1993	BSc(Eng), Environmental Engineering	University of Guelph, Guelph, ON

**RECENT PROJECT EXPERIENCE:**

**PRINCIPAL INVESTIGATOR – NSERC/HALIFAX WATER INDUSTRIAL RESEARCH CHAIR IN WATER QUALITY AND TREATMENT**

Clients: Halifax Water, LuminUltra Ltd., CBCL Ltd., Mantech Ltd., CBRM Water Department, AGAT Laboratories (2007- present)

Project Description: The NSERC / Halifax Water Industrial Research Chair is multifaceted research program that considers improvement of drinking water quality from source to tap. Through the Research Chair, this program has made considerable advancements in understanding and managing lead in drinking water.

**CO-INVESTIGATOR – *OPTIMIZING LEAD CONTROL CITY OF REGINA***

Client: City of Regina, SK (2017-18)

The overall goal of this project was to address corrosion control strategies for the City of Regina.

**TECHNICAL ADVISOR – *HEALTH CANADA – GUIDELINE DEVELOPMENT PROCESS***

Client: Health Canada (2016-19)

Dr. Gagnon through Armview Engineering was contracted to provide peer review and/or document preparation for Health Canada in preparing for drinking water guidelines. Example contributions include natural organic matter, aluminum, iron and microbiological safety have occurred recently.

**CO-PRINCIPAL INVESTIGATOR – OPTIMIZING BIOFILTRATION FOR IMPROVED MANGANESE CONTROL DURING COLD WEATHER SEASONS**

Client: Alameda County Water District Water Research Foundation (2017-2020)

Project Description: This project aims to understand biofilms in drinking water biofiltration and improve their efficacy for removing and managing manganese.

**SCIENTIFIC CONTRIBUTIONS:**

*Dr. Gagnon has published over 175 journal articles related to water treatment and over 300 conference papers. According to Google scholar, Dr. Gagnon's h-index is 37 with over 4100 citations. A selection of some of his contributions are provided below*

**Li, B. B.F. Trueman, S. Munoz, J.A. Loscin** and G.A. Gagnon. 2020. Impact of sodium silicate on lead release on colloid size distributions in drinking water. *Water Research*. Revisions requested.

**Dunnington, D., B. Trueman, W. Raseman, L. Anderson,** and G.A. Gagnon. 2020. Comparing the predictive performance, interpretability, and accessibility of machine learning and physically-based models for water treatment. *ACS ES&T Engrg*. Revisions requested.

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**Li, B., B. Trueman, Md. S. Rahman,** and G.A. Gagnon. 2020. Controlling lead release due to uniform and galvanic corrosion - an evaluation of silicate-based inhibitors. *J. Hazardous Materials*. Revisions requested

**Sweeney, C.L. J.L. Bennett, C.A.M. Brown,** N.W. Ross, and G.A. Gagnon. 2021. Validation of a QuEChERS method for extraction of estrogens from a complex water matrix and quantitation via high-performance liquid chromatography-mass spectrometry. *Chemosphere*. 128315.

**Maclsaac, S., C.L. Sweeney** and G.A. Gagnon. 2020. Instrument hacking: repurposing and recoding a multi-well instrument for automated, high-throughput monochromatic UV photooxidation of organic compounds. *ACS ES&T Engrg*. DOI 10.1021/acsestengg.0c00123

**DeMont, I.,** A.K. Stoddart and G.A. Gagnon. 2020. Assessing strategies to improve the efficacy and efficiency of direct filtration plants facing changes in source water quality from anthropogenic and climatic pressures. *J. Water Process Engrg*. 101689.

**Secka, F., N.E. Allward,** A.K. Stoddart and G.A. Gagnon. 2020. An automated and high-throughput method for adenosine triphosphate (ATP) quantification *AWWA Water Science*. DOI 10.1002/aww2.1202

Ontiveros, C. C., **C.L. Sweeney,** C. Smith, **S. Maclsaac, S. Munoz,** R. Davidson, C. McCormick, N. Thomas, I. Davis, A.K. Stoddart and G.A. Gagnon. 2020. Characterization of a commercially available, low pressure UV lamp as a disinfection system for decontamination of common nosocomial pathogens on N95 Filtering Facepiece Respirator (FFR) material. *Env. Sci.: Water Res. & Tech.* 6: 2089–2102

**Lane, K.** and G.A. Gagnon. 2020. Evaluating the use and intent of drinking water advisories in Atlantic Canada. *Water Policy*. DOI 10.2166/wp.2020.029

**Lane, K., B.F. Trueman, J. Locsin** and G.A. Gagnon. 2020. Inorganic contaminants in Canadian First Nation community water systems. *J. Water & Health*. DOI 10.2166/wh.2020.185

**Dunnington, D.W.,** S. Roberts, S.A. Norton, I.S. Spooner, J. Kurek, J.L. Kirk, D.C.G. Muir, C.E. White, and G.A. Gagnon 2020. The distribution and transport of two centuries of lead deposition as recorded by lake sediments from northeastern North America. *Science of Total Environment*. 140212.

**Gora, S. L.,** T. Anaviapik-Soucie, **N.E. Allward, C.C. Ontiveros,** V. L'Herault, J. Campbell, M. Gavin, **B.F. Trueman,** A.K. Stoddart and G.A. Gagnon. 2020. Microbiological water quality in a decentralized

Arctic drinking water system. *Env. Sci.: Water Res. & Technol.* 6(7): 1855-1868.

**Locsin, J.M., B.F. Trueman, D. Serracin-Pitti, G.M.L. Stanton** and G.A. Gagnon. 2020. Potential regulatory implications of guideline changes in lead monitoring for Nova Scotia. *AWWA Water Science*. 2(4) e1182.

**Dunnington, D.W., B.R.B. Gregory, I.S. Spooner, C.E. White** and G.A. Gagnon. 2020. Evaluating the performance of calculated elemental measures in sediment archives. *J Paleolimnol* 64(2): 155-166.

**Traver, E., R.A. Karaballi, Y.E. Monfared, H. Daurie, G.A. Gagnon & M. Dasog.** 2020. TiN, ZrN, and HfN nanoparticles on nanoporous aluminum oxide membranes for solar-driven water evaporation and desalination. *ACS Applied Nano Materials*. DOI 10.1021/acsanm.0c00107

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**Abkar, L., A.K. Stoddart** and G.A. Gagnon. 2019. Sedimentation: hydraulic improvement of biofiltration. *AWWA Water Science*. DOI: 10.1002/aws2.1160

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**Park, Y., A. Alayafi, J.L. Bennett, D. Malloy, P.M. Smith, N.W. Ross, G.A. Gagnon** and A.A. Donaldson. 2019 Modeling the fate of dietary 17B-estradiol and its metabolites in an American eel (*Anguilla rostrata*) recirculating aquaculture system. *Aquacult. Engineering* 86: 101995

**Li., B., B.F. Trueman, M.S. Rahman, Y. Gao, Y. Park** and G.A. Gagnon. 2019. Understanding the impacts of sodium silicate on water quality and iron oxide particles. *Environ. Sci.: Water Research & Tech.* 5: 1360-1370

**Trueman, B.F., B. Gregory, N. McCormick, Y. Gao, S.L. Gora, T. Anaviapik-Soucie, V. L'Hérault,** and G.A. Gagnon. 2019. Manganese increases lead release to drinking water. *Enviro. Sci. & Technol.* 53(9): 4803-4812.

**Brophy, M., A Mackie, Y. Park** and G.A. Gagnon. 2019. Exploring the detection of microcystin-LR using polar organic chemical integrative samplers (POCIS). *Environmental Science: Processes & Impacts*. 21: 659-666.

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**Gao, Y., B. Trueman, A.K. Stoddart** and G.A. Gagnon 2018. Understanding the impact of extracellular polymeric substances on lead release in drinking water systems. *ACS Omega*. 3(11): 14824-14832.

**Rauch, K., A. Mackie, B. Middleton, X. Xie** and G.A. Gagnon. 2018. Biomass recovery method for adenosine triphosphate (ATP) quantification following UV disinfection. *Ozone Sci & Engrg.* DOI:10.1080/01919512.2018.1518127

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## HONOURS AND AWARDS:

2020 Fellow, Canadian Society for Civil Engineering (CSCE). Awarded to CSCE members that have minimum of 10 years of professional experience and have made distinguishing contributions to Civil Engineering.

- 2019 Professional of Distinction, Nova Scotia Discovery Awards. The Discovery Awards recognize talented individuals in STEM in Nova Scotia. This award recognizes an individual who is a leading innovator and role model for people who choose science as a career.
- 2018 NSERC Discovery Accelerator Award. Each year, Natural Science & Engineering Research Council of Canada (NSERC) awards up to 125 Accelerator Award to highly rated STEM-based researchers. Each award will have a total value of \$120,000 to be paid over three years. Dr. Gagnon also received this award in 2008.
- 2016 University Research Professor, Dalhousie University. Dr. Gagnon is the first engineering professor at Dalhousie to be recognized with this award, which recognizes excellence in University research and scholarship
- 2014 George Fuller Award, ACWWA/AWWA. This award is in recognition Dr. Gagnon's distinguished service to the water supply field in commemoration of the sound engineering skill and the constructive leadership to the water industry.
- 2012 Fellow, Engineers Canada. Dr. Gagnon was honoured by Engineers Canada for his years of service in Engineers Nova Scotia.

#### PROFESSIONAL ORGANIZATIONS:

*Dr. Gagnon is a member of the following professional organizations. He has been a very active member of AWWA serving on in its Board of Directors between 2017-2020 and many of its technical committees.*

American Water Works Association (AWWA)  
American Chemical Society (ACS)  
Association of Environmental Engineering and Science Professors (AEESP)  
Canadian Society for Civil Engineering (CSCE)  
Canadian Association on Water Quality (CAWQ)  
Engineers Nova Scotia  
International Water Association (IWA)